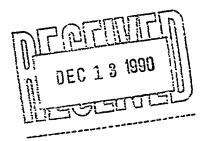# Y-12

## OAK RIDGE
## Y-12
## PLANT

**MARTIN MARIETTA**

Analysis of Proposed Postclosure
Alternatives for the Oil Landfarm
Waste Management Area at the
Oak Ridge Y-12 Plant,
Oak Ridge, Tennessee

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            Travis, D. A. White

Abstract:   Assessment of risks to human health and the environment posed by the
            Oil Landfill WMA under three remedial scenarios.  PCE, TCE, and
            1,1-DCE were determined to be the primary contaminants of concern.
            Groundwater is the most important migration pathway for the
            contaminants.  Transport via surface waters, air, and movement via
            soils and sediment are relatively unimportant.  Modeling predicts
            the VOC- contaminated groundwater is eventually discharged to Bear
            Creek.  Health risks are at unacceptable levels adjacent to the WMA.
            Includes waste disposal history, and groundwater and surface water
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yes, we have a copy.

Environmental Restoration Division
Y-12 Environmental Restoration Program

# Analysis of Proposed Postclosure Alternatives for the Oil Landfarm Waste Management Area at the Oak Ridge Y-12 Plant, Oak Ridge, Tennessee

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Date Issued—December 1990

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# EXECUTIVE SUMMARY

The Oil Landfarm Waste Management Area (WMA) is located in Bear Creek Valley about 1 mile southwest of the Y-12 Plant on the U.S. Department of Energy (DOE) Oak Ridge Reservation. From 1943 until 1982, several types of solid and liquid wastes were deposited in the five disposal areas that constitute the Oil Landfarm WMA. The disposal areas are: the Oil Landfarm disposal plots, the Boneyard, the Burnyard, the Chemical Storage Area, and the Sanitary Landfill. The Oil Landfarm disposal plots were used from 1973 until 1982 for the biological degradation of oily wastes. The Boneyard was active from 1943 to 1970 and received a great variety of wastes for burning or burial, including organics, metals, acids, and debris. The Burnyard is located beneath the present Chemical Storage Area. It operated from 1943 to 1968 and consisted of unlined trenches in which various wastes from plant operations were ignited with solvents or oils and burned. The Chemical Storage Area operated from 1975 to 1981 for the disposal of wastes that posed safety hazards; for example, reactive, corrosive, and explosive chemicals. Finally, the Sanitary Landfill was used from 1968 to 1980 for the burial of solid wastes. Although the Sanitary Landfill was supposed to receive only nonhazardous wastes, it may contain toxic chemicals and contaminated materials.

Since 1982 the Y-12 Plant has sampled groundwater, surface water, soils, and sediments in Bear Creek Valley. Data from this sampling program show that at the Oil Landfarm WMA groundwater is the most seriously contaminated medium. The chief contaminants of groundwater are the volatile organic compounds (VOCs). Concentrations of several VOCs are in the range of 100s to 1000s of $\mu$g/L in shallow wells adjacent to waste disposal areas. Elevated levels of some metals, gross alpha, and gross beta occur in isolated wells. Groundwater contamination currently extends only a few hundred feet from the site. Soils in the immediate vicinity of the disposal areas contain elevated levels of organic compounds, metals, and polychlorinated biphenyls (PCBs). Stream sediments at the site contain only slightly elevated levels of some metals. Surface waters at the Oil Landfarm WMA contain elevated levels of uranium and trace amounts of VOCs.

A network of monitoring wells is in place at the Oil Landfarm WMA, and groundwater assessment monitoring as prescribed under the Resource Conservation and Recovery Act (RCRA) is performed on a quarterly basis. In addition, several aquifer tests have been performed in the valley, providing much information on the hydrogeological system. The Y-12 Plant has also undertaken various pollution containment measures. The Sanitary Landfill was capped with clay and topsoil and revegetated in 1983. In 1989 large volumes of contaminated soils were excavated from the Oil Landfarm disposal plots, and the plots and the Chemical Storage Area have been covered with multilayer, engineered caps. Capping of the Boneyard is under way. Additional remedial actions are currently being evaluated as part of the Closure and Postclosure Activities (CAPCA) Project.

This study assesses the risks to human health and the environment posed by the Oil Landfarm WMA under three remedial scenarios:

1. take no action (reference case);

2.  place impermeable caps over the disposal areas to reduce leachate production and contaminant input to the aquifer; and

3.  in addition to the measures described in (2), construct a network of recovery wells to extract contaminated groundwater for treatment in an air-stripping facility.

Two cases are considered for the recovery well option: wells pumping over only the upper 200 ft of the aquifer and wells pumping over 600 ft of the aquifer. In addition, we attempt to estimate the effects of possible long-term contaminant sources on the capping and recovery alternatives.

Based on their potential for harm to the public health, three contaminants are chosen for risk assessment: tetrachloroethene (PCE), trichloroethene (TCE), and 1,1-dichloroethene (1,1-DCE). The potential for migration of these contaminants via transport in groundwater, surface water, air, and movement with soils and sediments is evaluated using computer modeling and qualitative arguments. The results of this analysis show that groundwater is the most important migration pathway for the contaminants. Transport in surface waters, air, and movement via soils and sediments are likely to be unimportant compared to transport in groundwater.

Because the VOCs have low water solubilities and are denser than water, there is a good possibility that a portion of the VOCs disposed of at the Oil Landfarm WMA are now in the saturated zone as nonaqueous phase liquids (NAPLs). If this is the case, the NAPLs can be expected to act as long-term sources of contaminants to the aquifer in spite of capping. Because it is not possible to predict the actual effectiveness of capping, two limiting capping scenarios are modeled: (1) fully effective caps, which eliminate sources of contaminants to the aquifer, and (2) completely ineffective caps, which do not decrease contaminant input to the aquifer. In the first case model, sources (source terms) are turned off during capping and recovery simulations. In the second case model, sources run throughout the simulations.

Because of the hydrogeological properties of the site, computer modeling predicts that the maximum extent of migration of the contaminants in groundwater under all scenarios is less than 2000 ft downgradient (SW) from the southwest corner of the WMA. Migration is limited largely because the plumes are predicted eventually to intersect Bear Creek and discharge contaminants into surface water. Capping does not significantly affect the extent of migration, and the 600-ft recovery well option has relatively little effect on this aspect of the contaminant plumes. The 200-ft recovery well option is only slightly more effective in limiting migration. Whether or not sources run throughout the simulations does not significantly affect the extent of plume migration.

Although the extent of migration is predicted to be limited, high concentrations of VOCs persist at the site for long times under all scenarios (i.e., about 40 years). Because there are no data from which to constrain source-term durations, no-action and capping-with-sources and recovery-with-sources computer simulations use constant source terms throughout the full 100 years of simulations as worst-case scenarios. Although this procedure almost certainly overestimates the persistence of the predicted plumes, plume areas and peak concentrations eventually reach steady-state conditions because of the discharge of contaminants to Bear

Creek. Without sources in the simulations, plume areas and peak concentrations produced by the capping simulations are significantly smaller than those from the no-action simulations (the upgradient boundaries of the plumes migrate away from their original sources), and the simulated plumes persist for 30 to 60 years. With source terms, the results of capping simulations are essentially the same as the no-action results, and the plumes persist indefinitely.

The presence or absence of sources in the recovery simulations does not change the relative efficacy of recovery with respect to capping or of recovery over the two depths considered. Either with or without sources, recovery in the 600-ft case is little more effective than capping in reducing contaminant concentrations and plume areas. Without sources, the 600-ft recovery plumes persist for about 30 to 60 years; with sources, they, of course, persist indefinitely. The 200-ft case, either with or without sources, is somewhat more efficient, and peak concentrations and plume areas are both reduced by about a factor of 2 with respect to the capping and 600-ft results at given times. Without sources, the plumes dissipate in about 25 to 40 years; with sources, they persist indefinitely. The recovery well networks are not very efficient because of restrictions on the placement of wells imposed by the proximity of Bear Creek, the cavity-ridden Maynardville formation and limitations on pumping rates in other formations. In addition, relatively slow transport velocities, especially for PCE, extend cleanup times.

A quantitative assessment of the risks to human health posed by groundwater was performed using predicted contaminant concentrations. Because of the limited extent of migration of the contaminant plumes under all scenarios, significant health risks occur only in the vicinity of the site. However, for all alternatives, health risks at the site are above acceptable levels for longer than 40 years. For no-source scenarios, capping reduces health risks significantly with respect to the no-action alternative, but the recovery well network scenarios are not much more effective than capping in reducing health risks. For scenarios with sources, health risks associated with capping are the same as for no action, but the recovery well scenarios are still not much more effective than capping in reducing risks.

As alluded to earlier, modeling predicts that VOC-contaminated groundwater eventually discharges to Bear Creek along the reach immediately south and southwest of the Oil Landfarm WMA. Because of dilution and loss of contaminants to air by volatilization, concentrations of contaminants in Bear Creek can be expected to be much lower than those in groundwater and to fall off rapidly downstream from the Oil Landfarm WMA. Because there are no data from which to calibrate them, calculations of future concentrations in Bear Creek were not attempted for this site. However, based on groundwater concentrations and previous experience, we would expect peak no-action concentrations in Bear Creek immediately south of the WMA to be of the order of 10s of $\mu g/L$, and peak concentrations for the other alternatives to be even less.

Under the recovery well scenarios, groundwater extracted from the Oil Landfarm WMA site is to be transported to the Bear Creek Burial Grounds, where it will be treated together with groundwater from that site in an air-stripping facility, the Groundwater Treatment Facility (GWTF). Operation of the GWTF would result in discharge of contaminants to the atmosphere. Air modeling predicts that concentrations of contaminants in air at the time of

GWTF startup (1997) from the combined treatment of water from the Burial Grounds and from the Oil Landfarm WMA will be above acceptable levels only in the immediate vicinity of the facility. The Oil Landfarm WMA's contribution to air contamination is relatively small compared to that of the Burial Grounds.

Large volumes of PCB- and VOC-contaminated soil have already been excavated from the Oil Landfarm disposal plots. Emplacement of caps over the disposal areas should essentially eliminate erosion of soils and subsequent transport on sediments as pathways for contaminant migration. The recovery well option would have little additional effect on migration via these pathways.

There is little to indicate that contaminant transport from this site has had an ecological impact in Bear Creek, where ecological conditions are dominated by the effects of inputs from the S-3 groundwater plume. Present and predicted future concentrations of contaminants in Bear Creek as a result of contributions from the Oil Landfarm WMA and other nearby disposal sites are not high enough to cause ecological harm.

Adverse ecological effects of the proposed remedial actions are likely to occur in Bear Creek. While such impacts are not likely to be large, they could represent a setback in the continuing ecological recovery of Bear Creek. Groundwater withdrawals will result in flow reductions and increased intermittent character in a 2.5-km reach downstream from the Oil Landfarm WMA. Slight decreases in aquatic habitat and stream productivity are likely. Because the toxicity of contaminants from the S-3 plume is reduced at downstream sites by inputs of less contaminated groundwater, reducing groundwater inputs may slightly exacerbate toxicity problems. Conversely, discharging treated groundwater to the headwaters of Bear Creek could act to ameliorate toxic groundwater inputs near the S-3 pond site.

Estimated total costs for capping the disposal areas (not including the Sanitary Landfill) of the Oil Landfarm WMA are $8 million. Operating and maintenance costs for the cap are expected to be minimal. No cost estimate has been made for the specific recovery well and treatment scenarios considered in this study. However, based on other studies, capital costs are expected to be about $10 to $20 million, in addition to capping costs. Expected operation and maintenance costs are approximately $2 million/year.

Our analysis of the alternatives just described leads to the following conclusions and recommendations:

1. Health risks associated with the Oil Landfarm WMA are confined to the immediate vicinity of the site.

2. Health risks under all alternatives can be expected to remain at unacceptable levels for more than 40 years at locations adjacent to the WMA.

3. The effect capping the disposal areas will have on contaminant input to groundwater is uncertain. If a significant portion of the contaminants at the site exist as NAPLs in the saturated zone, capping may not be very effective in reducing contaminant sources and ameliorating risks associated with groundwater at the site.

4. Recovery of contaminated groundwater with extraction wells is predicted to be little more effective than capping in reducing risk levels at this site. This conclusion is independent of whether or not long-term NAPL sources are present.

5. Several types of data are essential to more realistically evaluate the efficacy of the capping and recovery well scenarios. The locations and natures of the VOC sources need to be better characterized. The areal and vertical distributions of contaminants in groundwater need to be better understood to more fully evaluate recovery well scenarios, especially the depth over which contaminants should be recovered. Finally, more accurate data on the hydrologic properties of the formations at the site (e.g. sustainable yield per well and hydraulic conductivity as a function of depth) would be necessary for designing optimum recovery well networks or evaluating other alternatives.

6. Because exposure of the general public to contaminants from this site is unlikely in the foreseeable future, and because traditional groundwater recovery techniques are predicted to be relatively ineffective in reducing risk levels, we recommend that consideration be given to alternative remedial technologies (e.g., in situ biotreatment, steam displacement, and surfactants) rather than proceeding immediately with groundwater recovery and treatment. Realistically evaluating such alternative measures would require the same types of data outlined in (5) above.

# 1. INTRODUCTION

## 1.1 FACILITY DESCRIPTION AND SETTING

The Oil Landfarm Waste Management Area (WMA) is located in Bear Creek Valley about 1 mile southwest of the Y-12 Plant (Fig. 1.1). The plant was built by the U.S. Army Corps of Engineers in 1943 as part of the Manhattan Project. Its original mission was to separate $^{235}U$ from natural uranium by the electromagnetic process; that process was discontinued after World War II. The Y-12 Plant is now operated for the U.S. Department of Energy (DOE) by Martin Marietta Energy Systems, Inc. (Energy Systems) and has developed into a sophisticated design and manufacturing organization that (1) produces nuclear weapons components, (2) processes special materials, (3) supports other DOE installations, and (4) supports other government agencies.

Bear Creek Valley, which trends northeast-southwest, is bordered on the northwest by Pine Ridge and on the southeast by Chestnut Ridge. The crests of the ridges are a few hundred feet above the valley floor, which is a few thousand feet wide in most places. Bear Creek originates at the southwestern edge of the Y-12 Plant and follows a southwesterly course along the base of Chestnut Ridge until it discharges into East Fork Poplar Creek about 8 miles from its headwaters. Several small tributaries flow southward from Pine Ridge into Bear Creek and several springs on the north slope of Chestnut Ridge also discharge into the creek.

## 1.2 WASTE-DISPOSAL HISTORY AND CURRENT STATUS

Since 1943, wastes from the Y-12 Plant have been deposited in several sites located southwest of the plant along Bear Creek Valley (Fig. 1.2). The Oil Landfarm WMA includes five disposal areas: the Oil Landfarm disposal plots, the Boneyard, the Burnyard, the Chemical Storage Area, and the Sanitary Landfill. The Oil Landfarm disposal plots were used from 1973 until 1982 for the biological degradation of oily wastes. The Boneyard, located east of the Oil Landfarm disposal plots, was active from 1943 to 1970 and received a great variety of wastes for burning or burial, including organics, metals, acids, and debris. The Burnyard is located beneath the present Chemical Storage Area. It operated from 1943 to 1968 and consisted of unlined trenches in which various wastes from plant operations were burned. The Chemical Storage Area operated from 1975 to 1981 for the disposal of wastes which posed safety hazards (e.g., reactive, corrosive, and explosive chemicals). Finally, the Sanitary Landfill was used from 1968 to 1980 for the burial of solid wastes. Although the Sanitary Landfill was supposed to receive only nonhazardous wastes, it may contain toxic chemicals and contaminated materials.

Since 1982 the Y-12 Plant has sampled groundwater, surface water, soils, and sediments in Bear Creek Valley. Data from this sampling program show that groundwater at the Oil Landfarm WMA is the most seriously contaminated medium. The chief contaminants of groundwater are the volatile organic compounds (VOCs), but elevated levels of some metals, gross alpha, and gross beta occur in isolated wells. Groundwater contamination currently

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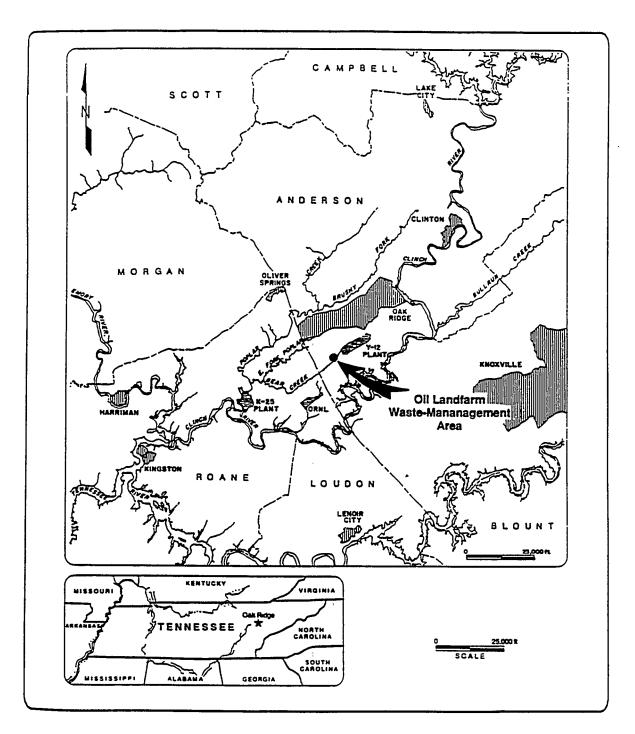
**Fig. 1.1. Location of the Oil Landfarm WMA** (Geraghty and Miller, Inc. 1989).
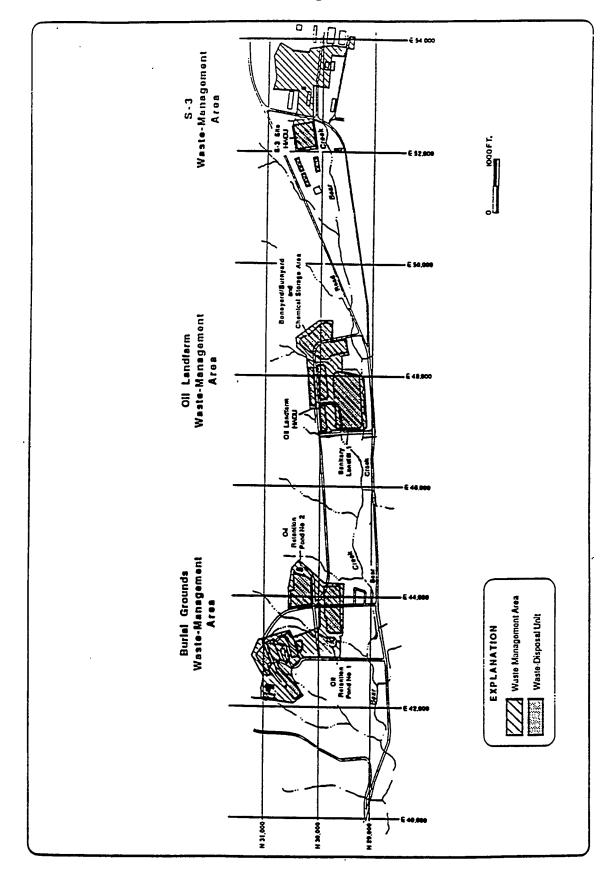
Fig. 1.2. Bear Creek Valley waste disposal area (Geraghty and Miller, Inc. 1989).

extends only a few hundred feet from the site. Soils in the immediate vicinity of the disposal areas contain elevated levels of organic compounds, metals, and polychlorinated biphenyls (PCBs). Stream sediments at the site contain only slightly elevated levels of some metals. Surface waters at the Oil Landfarm WMA contain elevated levels of uranium and trace amounts of VOCs.

An extensive network of monitoring wells now exists in Bear Creek Valley, and groundwater assessment monitoring as prescribed under the Resource Conservation and Recovery Act (RCRA) is performed on a quarterly basis. In addition several aquifer tests have been performed in the valley, providing much information on the hydrogeological system. The Y-12 Plant has also undertaken various pollution containment measures. The Sanitary Landfill was capped with clay and topsoil and revegetated in 1983. In 1989 large volumes of contaminated soils have been excavated from the Oil Landfarm disposal plots, and the plots and the Chemical Storage Area have been covered with multilayer, engineered caps. Capping of the Boneyard is under way. Additional remedial actions are currently being evaluated as part of the Closure and Postclosure Activities (CAPCA) Project.

## 1.3 OBJECTIVES OF THIS REPORT

This report assesses the risks to human health and the environment posed by the Oil Landfarm WMA under three remedial scenarios:

1. take no action (reference case),

2. place impermeable caps over the disposal areas to reduce leachate production, and

3. in addition to the measures described in (2), construct a network of recovery wells to extract contaminated groundwater for treatment in an air-stripping facility.

Two cases are considered for the recovery well option: wells pumping over only the upper 200 ft of the aquifer and wells pumping over 600 ft of the aquifer.

Sections 2–4 present background material on the geology and hydrology of the Oil Landfarm WMA site, waste-disposal practices at the site, and data on the present extent of contamination. Section 5 discusses the rationale for the choice of contaminants for quantitative risk assessment. Proposed remedial alternatives for the site are described in Sect. 6. Section 7 evaluates the migration and persistence of contaminants under each of the remedial scenarios. An assessment of the risk to human health posed by the site is presented in Sect. 8, and an assessment of ecological risk is presented in Sect. 9. Section 10 presents a summary of the findings of this study and makes recommendations for further action. Appendix A discusses the details of the groundwater modeling.

# 2. GEOLOGY AND HYDROLOGY

## 2.1 REGIONAL GEOLOGY

Bear Creek Valley is located in the Valley and Ridge physiographic province, which is bordered on the east by the Blue Ridge Mountains and on the west by the Cumberland Plateau. The province is characterized by a series of subparallel valleys and ridges that trend northeast-southwest. This regional structural trend is a result of major thrust faulting during Alleghenian time (approximately 260 to 340 million years ago), which folded and faulted the formations of the region. Paleozoic sedimentary formations underlie the area: the valleys are usually floored by shales and limestones; dolostones and sandstones are the ridge formers.

## 2.2 GEOLOGY OF THE OIL LANDFARM WMA SITE

The geology of Bear Creek Valley has been described in detail in previous reports (Geraghty and Miller, Inc. 1985; Bechtel National, Inc. 1984a; Bechtel National, Inc. 1984b; Rothschild 1984; King and Haase 1987). This section summarizes material presented in those reports.

### 2.2.1 Bedrock Formations

Bear Creek Valley at the Oil Landfarm WMA site is underlain by the Cambrian Conasauga Group (Fig. 2.1). The rocks of the Conasauga Group consist of layers and laminations of shale, siltstone, and limestone that have been divided into six formations on the basis of distinctive lithologic features (for detailed lithologic descriptions see King and Haase 1987). From oldest to youngest these formations are the Pumpkin Valley Shale, Rutledge Limestone, Rogersville Shale, Maryville Limestone, Nolichucky Shale, and the Maynardville Limestone (Fig. 2.2). The Conasauga Group rocks generally strike about N55°E and dip to the southeast. Dip angles range from 30° to near vertical. Small-scale folds with axes parallel to the strike of bedding are common in the Conasauga and contribute to the variability of bedding plane orientations.

The rocks of the Conasauga Group are highly fractured, with bedding plane partings as the principle fracture set. They also contain several joint sets at various orientations, with spacings from inches to several feet. Joints, bedding planes, and other zones of weakness in carbonate rocks provide preferred avenues for groundwater flow and for dissolution. The latter process is especially important in the Maynardville Limestone, and a high concentration of solution cavities underlies Bear Creek which lies in that formation. Dissolution also occurs in the other formations of the Conasauga but is less significant.

### 2.2.2 Unconsolidated Material

The unconsolidated material which overlies bedrock at the Oil Landfarm WMA site is composed primarily of weathered bedrock and man-made materials. Alluvium and colluvium occur along the instream channels and at the base of the steeper slopes in the valley. The depth to unweathered bedrock varies widely due to the different thickness of fill and alluvium
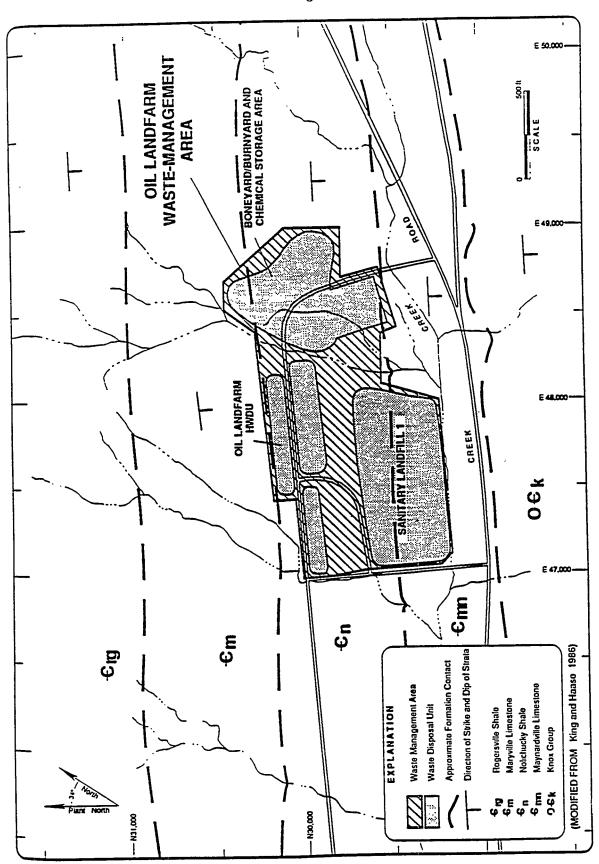
**Fig. 2.1. Bedrock geology in the vicinity of the Oil Landfarm WMA (Geraghty and Miller, Inc. 1989).**

| AGE | GROUP | LITHOLOGY | FORMATION |
|---|---|---|---|
| CAMBRIAN | KNOX | | COPPER RIDGE €cr |
| | CONASAUGA | | MAYNARDVILLE LIMESTONE €mn |
| | | | NOLICHUCKY SHALE €n |
| | | | MARYVILLE LIMESTONE €m |
| | | | ROGERSVILLE SHALE €rg |
| | | | RUTLEDGE LIMESTONE €rt |
| | | | PUMPKIN VALLEY SHALE €pv |
| | | | ROME €rm |

EXPLANATION

DOLOSTONE

LIMESTONE

LIMESTONE WITH INTERBEDDED SHALE

SANDSTONE WITH INTERBEDDED SHALE

SHALE

SHALE WITH INTERBEDDED LIMESTONE

Fig. 2.2. Stratigraphic column of bedrock units in Bear Creek Valley (Geraghty and Miller, Inc. 1988).

and the weathering characteristics of the underlying bedrock, but typically ranges from about 10 to 30 ft.

## 2.3 HYDROLOGY

The hydrology of Bear Creek Valley has been discussed in several previous reports. Geraghty and Miller, Inc. (1985), Bechtel National, Inc. (1984a), and Bechtel National, Inc. (1984b) present overall reviews of the regional hydrology. Surface water flow data for Bear Creek and the tributaries is contained in a report by Martin Marietta Energy Systems, Inc. (1985a). Reports on aquifer tests which provide measured values for several aquifer parameters are discussed and referenced in Appendix A in the description of groundwater modeling.

### 2.3.1 Groundwater System

The hydrogeologic system underlying Bear Creek Valley is essentially a single aquifer of low water-transmitting capacity. Hydraulic conductivity decreases with depth, but there is no sharp discontinuity in this parameter between the unconsolidated zone and the bedrock units of the Conasauga Group. Where solution cavities are abundant, as in the Maynardville Limestone, measured hydraulic conductivities are at least an order of magnitude higher than in other formations. Measured values range from 263 ft/year in a shallow, cavity-ridden zone at the contact between the Maynardville Limestone and the Nolichucky Shale to 0.01 ft/year in zones deeper than 450 ft in the Nolichucky.

The permeability of the bedrock units is due primarily to the fracture and joint systems described in the previous section rather than to intergranular porosity. As a result the aquifer is highly anisotropic, with preferred flow directions along strike and dip. Pump tests show that permeabilities parallel to bedding are as much as an order of magnitude greater than those normal to bedding. The joint sets and solution cavities may also provide avenues for flow which are superimposed on the overall anisotropy. The aquifer is, in addition, heterogeneous: not only does permeability vary with depth, but it also varies within and across the different formations of the Conasauga Group.

The general direction of groundwater flow in the vicinity of the Oil Landfarm WMA is to the south, downslope and toward Bear Creek (Fig. 2.3). Water from precipitation which infiltrates the ground is eventually discharged to Bear Creek and the solution cavity system which underlies it. Water level contours for Feb. 1, 1989, are shown in Fig. 2.4 (Geraghty and Miller 1989). Although there are seasonal fluctuations in the water table, variations tend to be uniform across the area, and hydraulic gradients remain relatively unchanged. Hydraulic gradients across the area range from about 0.05 north of the Oil Landfarm disposal areas to about 0.01 along the trace of Bear Creek. If the aquifer were isotropic, groundwater flow would be normal to the contour lines; however, the overall anisotropy of the aquifer and the probable existence of preferred flow paths imply that substantial deviations from the ideal situation may occur. In addition, vertical components (both upward and downward) to flow are indicated in data from monitoring well clusters.
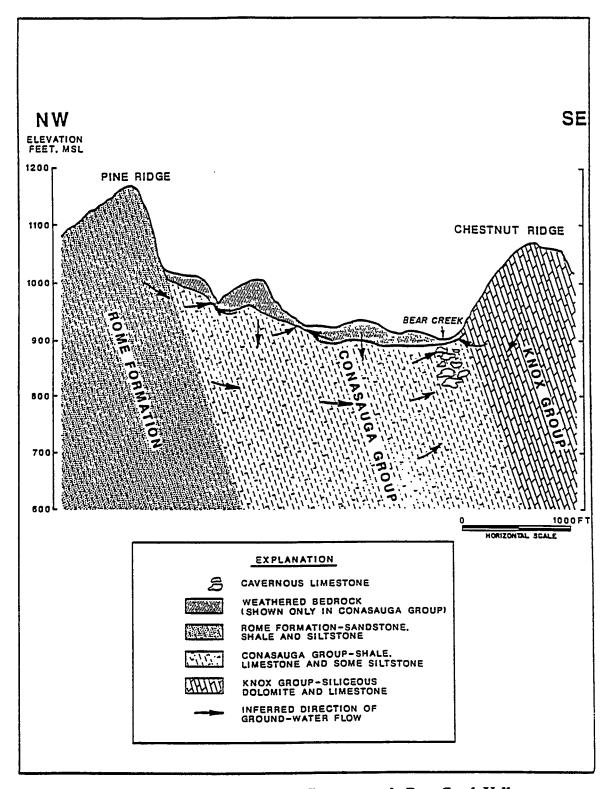
**Fig. 2.3. Generalized groundwater flow pattern in Bear Creek Valley (Geraghty and Miller, Inc. 1988).**
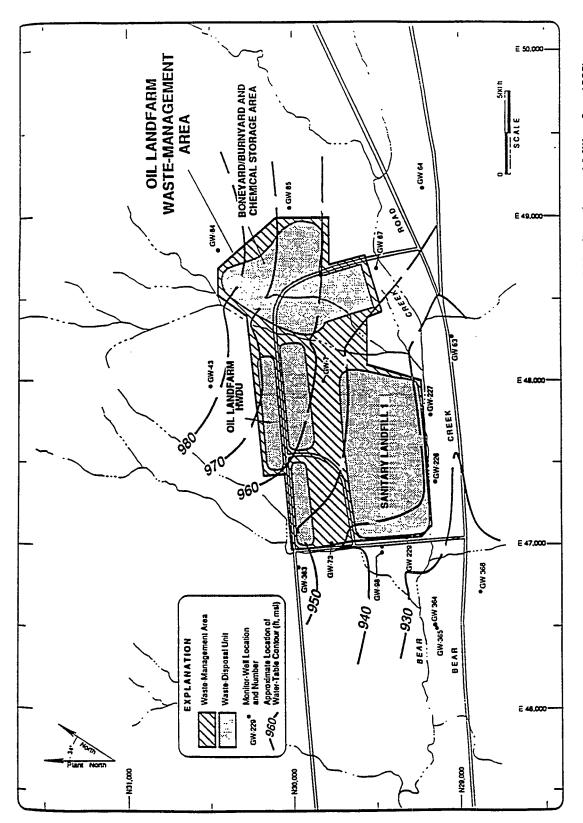
Fig. 2.4. Water table elevation contours at the Oil Landfarm WMA for Feb. 1, 1989 (Geraghty and Miller, Inc. 1989).

## 2.3.2 Surface-Water System

From its headwaters near the S-3 Ponds, Bear Creek flows southwest along the base of Chestnut Ridge, south of the Oil Landfarm WMA (Fig. 2.5). A few miles southwest of the Oil Landfarm WMA it turns north and flows into East Fork Poplar Creek, which discharges into the Clinch River. In the vicinity of the Oil Landfarm WMA, two tributaries, numbered 3 and 4 and referred to as NT3 and NT4, flow southward from Pine Ridge and discharge into Bear Creek, and two springs discharge to Bear Creek from Chestnut Ridge.

In general, base flow in Bear Creek and the tributaries is sustained by groundwater discharge. The highest flows in Bear Creek coincide with the times of greatest precipitation, usually the spring. The reach of Bear Creek south of the Oil Landfarm WMA is losing part of the year, and surface water and groundwater discharge into the cavity system below the creek. The outlets of the cavity system are not known, but there is evidence that at least some of the flow re-emerges downstream or through springs. When precipitation is low the tributaries and the reach of Bear Creek to the south of the Oil Landfarm may be dry.

Stream gaging sites along Bear Creek and the tributaries are designated using the Bear Creek Kilometer (BCK) system. The number following BCK indicates the location of the station in kilometers from the mouth of Bear Creek. Stations located on tributaries north and south of Bear Creek are given the modifiers NO and SO, respectively. The distance from the main stem of Bear Creek, in kilometers, follows the modifier.
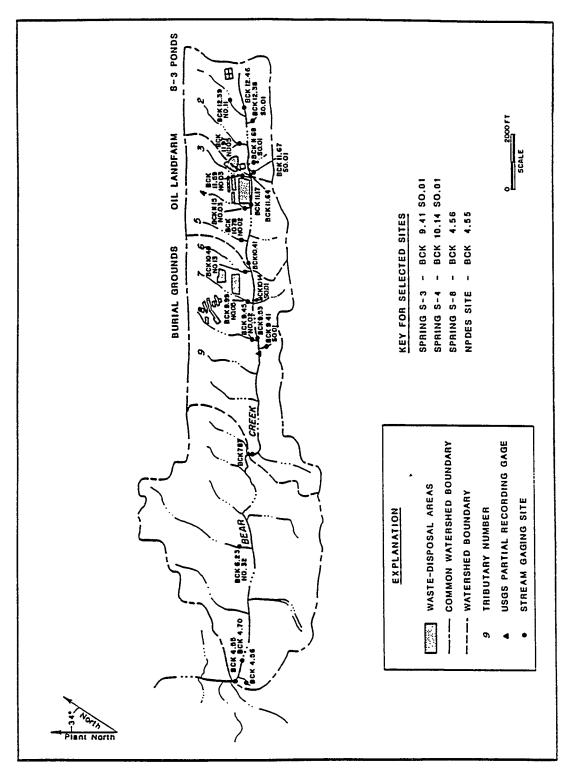
Fig. 2.5. Bear Creek watershed and locations of stream gaging sites (Geraghty and Miller, Inc. 1987c).

# 3. WASTE DISPOSAL HISTORY

The Oil Landfarm WMA includes five disposal areas: the Oil Landfarm disposal plots, the Boneyard, the Burnyard, the Chemical Storage Area, and the Sanitary Landfill (Fig. 3.1). A description of the disposal history of each of the areas is presented below. The descriptions are summarized from material in Geraghty and Miller, Inc. (1985), Union Carbide Corporation (1984), Bechtel National, Inc. (1983), and Geraghty and Miller, Inc. (1987c).

## 3.1 OIL LANDFARM

The Oil Landfarm disposal plots were used for the biological degradation of oily wastes from 1973 to 1982. Initially the site consisted of 1.3 acres but was gradually expanded to encompass 4.0 acres by 1979. Oily wastes were collected in tank trucks, spread over the surface of the plots, and plowed into the top 3 in. of soil. Plots were cultivated frequently between applications to enhance microbial activity. The facility was operated only during the dry months of the year (April to October). Approximately one million gallons of oily wastes were applied to the plots during the period of operation. Both the composition and volume of liquid waste applied varied from plot to plot. Wastes deposited at the Oil Landfarm included waste oils, coolants, mop waters, tanker oils from K-25, oil skimmed from the Oil Retention Ponds, and miscellaneous liquid wastes. The oils and coolants are known to have been contaminated with beryllium compounds, depleted uranium, PCBs, tetrachloroethene (PCE), and 1,1,1-trichloroethane.

## 3.2 BONEYARD

The Boneyard, located east of the Oil Landfarm disposal plots, was active from 1943 to 1970 and received a great variety of wastes, including organics, metals, acids, and debris. Most of the Boneyard was used for burial of construction spoil material. In the southwest corner of the Boneyard, however, unlined trenches were used to burn magnesium chips, which were ignited with solvents. The residue from the burning was covered with soil, compacted, covered again with topsoil, and seeded with grass. No collection or treatment systems were used at this site.

## 3.3 BURNYARD

The Burnyard is located beneath the present Chemical Storage Area. It operated from 1943 to 1968 and consisted of two unlined trenches in which various wastes were burned. The site is estimated to have received about 4000 tons per year of sanitary refuse from plant operations, including solids, liquids, and sludge. The wastes may have included empty pesticide containers, metal shavings, solvents, oils, and lab chemicals. Oils and other flammable liquids were used to ignite the waste materials. The trenches were eventually filled with soil. No collection or treatment systems were used at the site.

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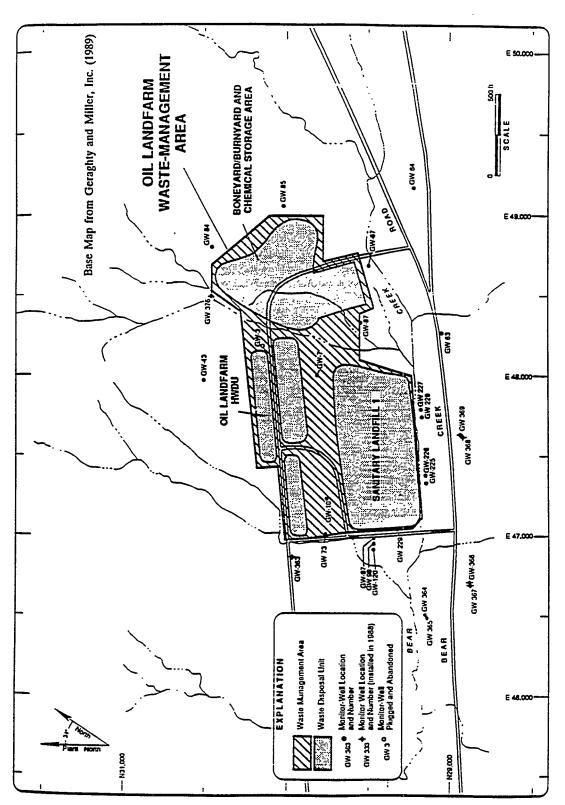


Fig. 3.1. Locations of Oil Landfarm WMA waste disposal areas (Geraghty and Miller, Inc. 1989).

## 3.4 CHEMICAL STORAGE AREA

The Chemical Storage Area operated from 1975 to 1981 for the disposal of wastes which posed safety hazards within the Y-12 Plant (e.g., reactive, corrosive, flammable, or explosive chemicals). Most of the waste came from two sources: gas cylinders with leaking or damaged valves, and lab chemicals. Less than 5 tons per year of wastes are estimated to have been disposed of at this site. Gasses from the cylinders were either bled directly to the atmosphere or through neutralizing filters, and the cylinders were either destroyed or sent for repair. Lab chemicals were exploded or reacted in a concrete vessel open to the atmosphere. The effluent from the vessel was discharged to a surface impoundment and allowed to percolate through the soil. Chemical residue remaining in the vessel was removed and buried elsewhere.

## 3.5 SANITARY LANDFILL

The Sanitary Landfill was used from 1968 to 1980 for the burial of solid wastes such as paper, plastics, animal bedding, and organic garbage. The landfill received approximately 105,000 tons of refuse during its period of operation. Although the Sanitary Landfill was supposed to receive only nonhazardous wastes, it is possible that it contains toxic chemicals and contaminated materials. The landfill was closed in 1983 by capping with 2 ft of clay and topsoil, and revegetating.

# 4. CURRENT CONTAMINATION

## 4.1 INTRODUCTION

Groundwater, surface water, sediments, and soils in the vicinity of the Oil Landfarm WMA are contaminated with one or more of the following contaminants: VOCs, heavy metals, PCBs, and radionuclides. Groundwater is the most seriously contaminated medium, and the VOCs are the chief contaminants of groundwater. Currently, relatively low levels of groundwater contamination extend along the trace of Bear Creek to about 500 ft from the southwest edge of the WMA. Soils in the immediate vicinity of the disposal areas contain elevated levels of organic compounds, metals, and PCBs. Stream sediments at the site contain only slightly elevated levels of some metals. Surface waters at the Oil Landfarm WMA contain trace amounts of organic compounds and elevated levels of uranium.

## 4.2 GROUNDWATER CONTAMINATION

Groundwater quality data have been collected at the Oil Landfarm WMA since 1983 (Bechtel National, Inc. 1983; Union Carbide Corporation 1984; Geraghty and Miller, Inc., 1985; Geraghty and Miller, Inc. 1987a; Geraghty and Miller, Inc. 1988; Geraghty and Miller, Inc. 1989; Martin Marietta Energy Systems, Inc. 1985b; Martin Marietta Energy Systems, Inc. 1987), and an extensive network of monitoring wells now exists at the site. Data on the locations, screened depths, and sampling histories of monitoring wells in existence by 1986 can be found in Geraghty and Miller, Inc. (1985; 1987c). Assessment monitoring is currently conducted in accordance with the *RCRA Ground-Water Quality Assessment Plan* (GWQAP), which was approved by the Tennessee Department of Health and the Environment (TDHE) in 1987 (Geraghty and Miller, Inc. 1987b). Figure 4.1 shows the assessment well network for 1988, which includes several wells installed since 1986, and Table 4.1 shows the screened intervals of the wells. The parameters that were sampled in 1988 are listed in Table 4.2.

Groundwater is the most seriously contaminated medium at the Oil Landfarm WMA. The description of groundwater contamination which follows is based primarily on data reported in the *1988 Ground-Water Quality Assessment Report* (Geraghty and Miller, Inc. 1989) but takes into account data from earlier sampling periods. The chief contaminants in groundwater are the VOCs. Concentrations of several VOCs are in the range of 100s to 1000s of $\mu$g/L in shallow wells adjacent to waste disposal areas. Table 4.3 shows the range of concentrations of contaminants measured in representative wells in 1988 and relevant drinking water standards for comparison. The compounds listed in Table 4.3 make up over 90% of the total VOCs measured at assessment wells.

As shown in Table 4.3, the areal distribution is not the same for different compounds occurring at the site. Distribution patterns suggest at least three separate source areas of VOCs at the site. The contaminants 1,1,1-trichloroethane, 1,1-dichloroethane, and 1,1-dichloroethene (1,1-DCE) are found in high concentrations only in shallow well GW-3, and these compounds and vinyl chloride are the only VOCs occurring in that well. Of the wells sampled in 1988, only GW-87 and GW-7 show high concentrations of PCE and *trans*-1,2-dichloroethene. High concentrations of trichloroethene (TCE) also occur in GW-7 and
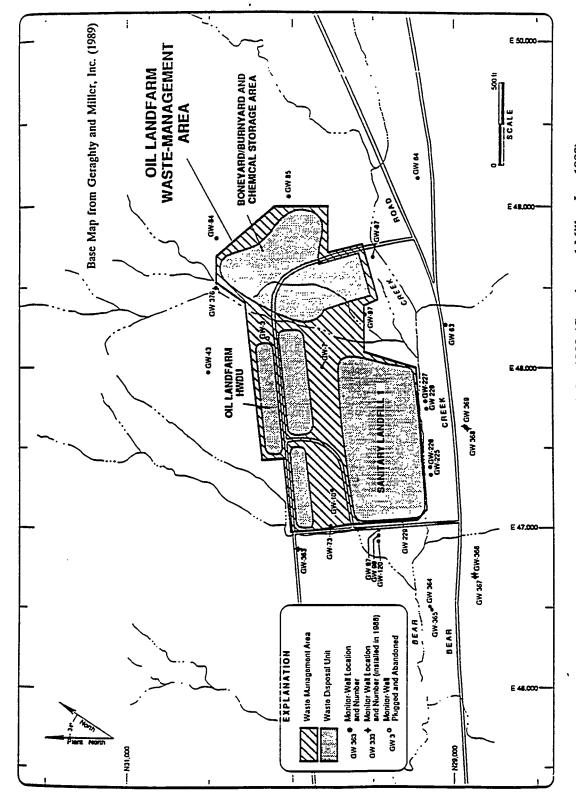
Fig. 4.1. Assessment monitor well network for 1988 (Geraghty and Miller, Inc. 1989).

GW-87. In addition, high levels of TCE, but no other VOC, occur in several wells located in the Maynardville Limestone along the trace of Bear Creek. Figures 4.2–4.4 illustrate these distribution patterns for 1,1-DCE, PCE, and TCE, respectively. The spatial distribution of TCE illustrated in Fig. 4.4 indicates that the primary source of TCE contamination in the wells in the Maynardville is upgradient from the Oil Landfarm WMA.

Table 4.1. Aquifer zones monitored by Oil Landfarm WMA assessment wells

| Well No. | Total depth (ft) | Screened/open interval (ft) | Aquifer zone |
|---|---|---|---|
| GW-3 | 35.2 | 18.0 – 28.0 | BDR |
| GW-7 | 16.5 | 12.2 – 14.3 | UNC |
| GW-10 | 15.0 | 7.7 – 12.7 | UNC |
| GW-43 | 40.0 | 22.8 – 32.8 | UNC |
| GW-63 | 35.0 | 27.7 – 32.7 | BDR |
| GW-64 | 57.0 | 46.8 – 52.7 | BDR |
| GW-67 | 16.5 | 11.2 – 16.2 | UNC |
| GW-73 | 81.0 | 69.8 – 79.8 | BDR |
| GW-84 | 34.0 | 22.8 – 27.8 | UNC |
| GW-85 | 62.0 | 48.4 – 58.8 | BDR |
| GW-87 | 19.0 | 9.0 – 19.0 | UNC |
| GW-97 | 19.2 | 11.8 – 16.8 | UNC |
| GW-98 | 104.0 | 82.4 –103.4 | BDR |
| GW-120 | 180.0 | 130.0 –180.0 | BDR |
| GW-225 | 200.0 | 150.0 –200.0 | BDR |
| GW-226 | 55.0 | 45.0 – 55.0 | BDR |
| GW-227 | 40.0 | 30.0 – 40.0 | BDR |
| GW-228 | 100.0 | 80.0 –100.0 | BDR |
| GW-229 | 55.0 | 40.0 – 55.0 | BDR |
| GW-363 | 75.0 | 50.0 – 75.0 | BDR |
| GW-364 | 60.3 | 47.0 – 60.3 | BDR |
| GW-365 | 150.0 | 126.7 –150.0 | BDR |
| GW-366 | 102.5 | 88.0 –101.0 | BDR |
| GW-367 | 151.4 | 125.0 –150.0 | BDR |
| GW-368 | 245.0 | 225.0 –245.0 | BDR |
| GW-369 | 150.2 | 115.8 –150.2 | BDR |

BDR = Bedrock zone.
UNC = Unconsolidated zone.

*Source:* Geraghty and Miller, Inc. 1988.

The vertical distribution of VOCs is difficult to assess from the available data. Except for TCE the highest levels of VOCs measured in 1986, 1987, and 1988 occur in wells less than 30 ft deep. However, two lines of evidence indicate that contamination extends, at least locally, to 245 ft (the depth of the deepest well at the site) or deeper. First, sampling in 1984 of GW-75 (which is screened in the Nolichucky at approximately 200 ft) showed concentrations of PCE and TCE comparable to those in adjacent shallow well GW-7.

Second, in several well clusters in the Maynardville, the deep wells consistently exhibit TCE concentrations comparable to or higher than those seen in the shallow wells.

Table 4.2. Groundwater quality parameters for the
1988 quarterly and annual assessments

---

*Metals*

| Lead | Beryllium |
| Barium | Uranium |
| Cadmium | Chromium |

*Major anions and cations*

| Nitrate (as N) | Alkalinity |
| Calcium | Potassium |
| Chloride | Sodium |
| Magnesium | Sulfate |

*Volatile organic compounds*

| Benzene | Chloromethane |
| trans-1,2-Dichloroethene | Methylene chloride |
| Carbon tetrachloride | 1,1,2,2-Tetrachloroethane |
| Chloroethane | Tetrachloroethene |
| 1,1-Dichloroethane | Chloroform |
| 1,2-Dichloroethane | 1,1-Dichloroethene |
| 1,1,1-Trichloroethane | 1,1,2-Trichloroethane |
| Trichloroethene | Vinyl chloride |

*Radiochemical Parameters*

| Gross Alpha | Gross Beta |

*Miscellaneous Parameters*

| pH | Specific Conductance |
| Total Dissolved Solids | Total Suspended Solids |

---

*Source:* Geraghty and Miller, Inc. 1989.

Data for selected compounds for sampling done from 1984 through 1988 were examined to look for time dependencies in measured concentrations. In general, concentrations of VOCs in a given well fluctuate widely from quarter to quarter and from year to year with no apparent pattern. A trend was discernible for only one well, GW-7. Concentrations of PCE and TCE in this well clearly decrease with time, indicating decreasing source concentrations.

Other contaminants of groundwater are trace metals and unidentified radionuclides. Measured concentrations of metals also vary widely from well to well and for different sampling periods. Barium, lead, chromium, uranium, and cadmium are reported as slightly above drinking water standards in at least one well for at least one sampling event in 1988.

Gross alpha and gross beta activity at levels above federal standards are observed in samples from several wells. The source of this activity is not presently known; several radionuclides were analyzed for in 1988 to help determine its origin, but the data were inconclusive. Additional monitoring for radionuclides was done in 1989, but the findings were not available for this report.

Table 4.3. 1988 Groundwater concentrations at selected
monitoring wells and relevant standards

| Contaminant | Concentration range (μg/L) | | | | Standard (μg/L) |
|---|---|---|---|---|---|
| | GW-3 | GW-7 | GW-87 | GW-225 | |
| Tetrachloroethene | G | 64–77 | 360–430 | <5–6 | 5[a] |
| Trichloroethene | G | 14–17 | 170–460 | 11–400 | 5[b] |
| trans-1,2-Dichloroethene | G | 130–170 | 110–150 | UD | 70[c] |
| 1,1-Dichloroethene | 620–620 | 4–4 | G–9 | <5–7 | 7[b] |
| 1,1,1-Trichloroethane | 240–510 | 2–2· | 4–7 | <5–3 | 200[b] |
| 1,1-Dichloroethane | 1900–2000 | 7–8 | 6–8 | <10–<10 | — |
| Vinyl Chloride | N–9 | 3–4 | N–3 | <10–<10 | 2[b] |
| Benzene | G | G | 26–39 | <5–<5 | 5[b] |
| Barium | 340–500 | 120–140 | 110–170 | 170–440 | 1000[b] |
| Lead | 10–11 | 46–160 | 12–29 | <4–20 | 50[b] |
| Cadmium | <2–<2 | 5–20 | <2–3 | <2–<2 | 10[b] |
| Uranium | <1–<1 | <1–<2 | 140–495 | <1–3 | 35[d] |
| Chromium | 10–80 | <10–30 | <1–11 | <10–<10 | 50[b] |
| Gross Alpha (pCi/L) | 5–9 | <1–<1 | 30–132 · | 6–6 | 15[b] |
| Gross Beta (pCi/L) | 5–64 | <1–11 | 88–165 | 10–46 | 50[c] |

*Source:* Concentration data from Geraghty and Miller, Inc. (1989).
G = Detection limit 5 μg/L.
N = Detection limit 10 μg/L.
UD = Unreliable data.
— = No standard available.
[a]Safe Drinking Water Act Proposed Maximum Contaminant Level.
[b]Safe Drinking Water Act Maximum Contaminant Level.
[c]Safe Drinking Water Act Proposed Maximum Contaminant Level Goal.
[d]Suggested No Adverse Affect Level for chronic ingestion.
[e]Based on 40 CFR Part 141.26 standard of 4 mrems per year.

Concentrations of nitrate above the drinking water standard of 10 mg/L occur in GW-85 and in several wells located in the Maynardville along the trace of Bear Creek. The occurrences in the Maynardville are part of the nitrate plume emanating from the former S-3 Ponds. The nitrate in GW-85 may be related to the S-3 plume or may represent an isolated nitrate source in the Oil Landfarm WMA.

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Fig. 4.2. Annual 1988 1,1-dichloroethene concentrations in selected monitor wells. Concentrations in μg/L.

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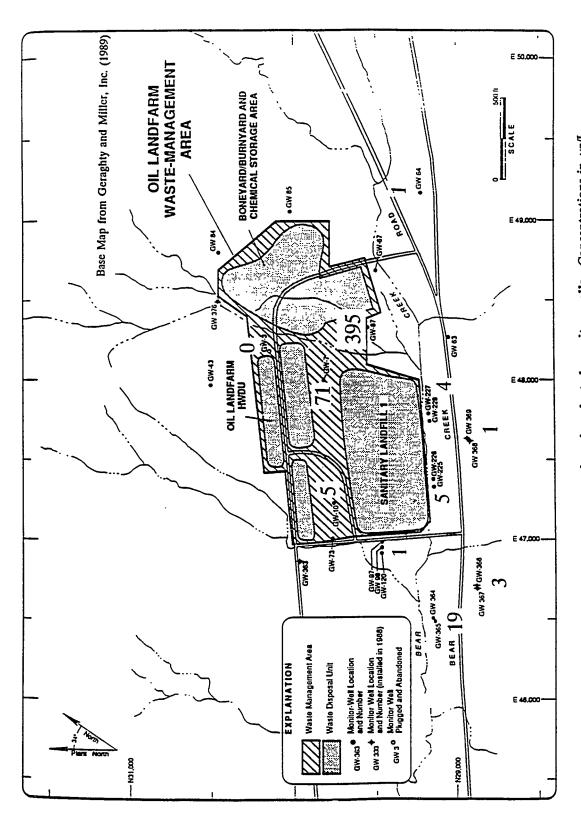


Fig. 4.3. Average 1988 PCE concentrations in selected monitor wells. Concentrations in $\mu g/L$.
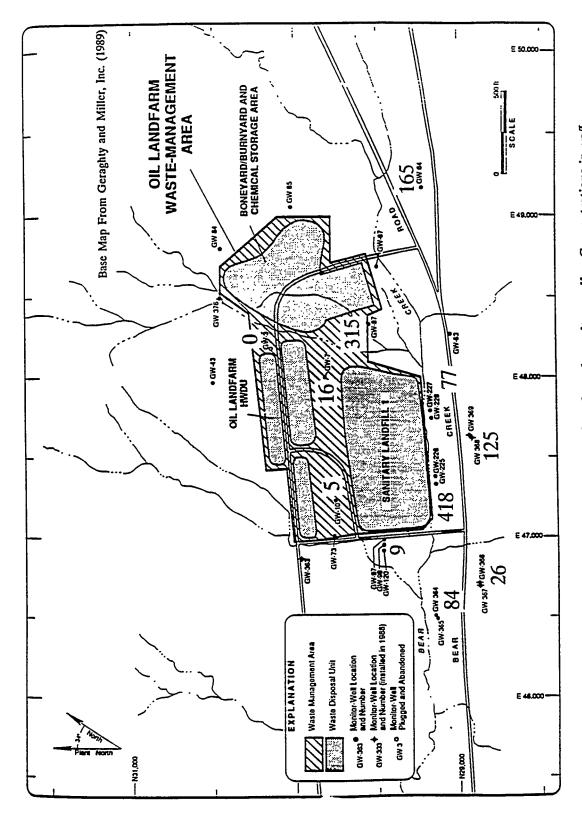
Fig. 4.4. Average 1988 TCE concentrations in selected monitor wells. Concentrations in $\mu g/L$.

## 4.3 SURFACE WATER, SEDIMENT, AND SOIL CONTAMINATION

Data for several contaminants in surface water and sediments in the vicinity of the Oil Landfarm WMA were collected from 1983 through 1986 by Bechtel National, Inc., and Roy F. Weston Laboratory (Martin Marietta Energy Systems, Inc. 1985b; Union Carbide Corporation 1984; Bechtel National, Inc. 1983; Bechtel National, Inc. 1984c; Roy F. Weston Laboratory 1986). More recent data for metals in Bear Creek water were obtained in sampling events conducted in March 1988 (Southworth 1988).

Surface waters in the vicinity of the Oil Landfarm WMA contain only trace amounts of organic compounds. Measured concentrations of VOCs in water in NT3 and NT4 and Bear Creek south of the Oil Landfarm WMA are reported as less than 10 $\mu$g/L for all sampling events since 1984. Sampling done in 1984 shows uranium at levels above background in NT3, and subsequent sampling events show elevated levels of uranium (above the suggested no adverse response level of 35 $\mu$g/L) along the reaches of Bear Creek both above and below the Oil Landfarm WMA. This distribution indicates that the site is not the only contributor of uranium to the creek. No other metal occurs in surface waters at levels above its drinking water standard. PCBs in water are reported as less than 0.3 parts per billion (ppb) in the tributaries for all aroclors for the 1984 sampling events, and no PCBs are reported in Bear Creek south of the Oil Landfarm WMA for later sampling periods.

The 1984 data show only trace amounts of VOCs in sediments in NT4, and data from later sampling events show no VOCs in Bear Creek sediments. Levels of uranium, mercury, titanium, and beryllium above background were found in the sediments of tributaries in the 1984 sampling. The highest measured values for mercury and beryllium, 7.5 parts per million (ppm) and 7.3 ppm, respectively, are well below health-based screening values. The highest measured values for titanium and uranium are 730 ppm and 635 ppm, respectively; there are no screening values with which to compare these concentrations. Later sampling events show uranium concentrations of the order of 100 to 300 ppm in the sediments in Bear Creek both above and below the Oil Landfarm WMA. Measured summed PCB concentrations in the sediments of the tributaries and Bear Creek are <1 ppm for all sampling periods, except for one location in NT4 at which one aroclor was 1.1 ppm in 1984.

The primary source of data on most contaminants in soils is sampling done in 1983 and 1984, the results of which are reported in Bechtel National, Inc. (1983), and Union Carbide Corporation (1984). Additional data for PCBs, oil and grease, and uranium in soils at the Oil Landfarm WMA were obtained in 1985 and 1986 and are reported in Herbes (1989). The primary contaminants measured in soil core samples taken in 1983 from wells adjacent to disposal areas (GW-1, GW-4, GW-5, GW-7, GW-11, and GW-13) and their highest concentrations (mg/kg) were: toluene (1.6), methylene chloride (0.96), PCE (1.1), and trans-1,2-dichloroethene (0.40). Composite soil samples taken in 1983 from the Oil Landfarm disposal plots showed the following major contaminants and maximum levels (in mg/kg): PCE (8.4), TCE (0.75), trans-1,2-dichloroethene (2.7), 1,1,1-trichloroethane (0.50), and several base-neutral and acid extractable organics. Concentrations of all contaminants were highest in the southeast group of plots. Data in Herbes (1989) indicate that PCBs, uranium, and oil/grease contamination are confined to the upper 6 in. of soil. The highest measured PCB concentration was 60 mg/kg, and mean concentrations of PCBs exceeded 25 mg/kg (the action level established for the Y-12 Disposal Area Remedial Action Project) in only two areas located in the southeast group of plots. The highest mean uranium concentration was

204 mg/kg, also in the southeast plots. In the Herbes (1989) study, oil/grease and PCB (but not uranium) concentrations decreased significantly with time, suggesting the influence of microbial degradation or volatilization on contaminant levels.

# 5. INDICATOR CHEMICALS

## 5.1 INTRODUCTION

Initial screening of data for environmental media at the Oil Landfarm indicates several contaminants are present in groundwater, soils, surface water, and sediments. Performing a quantitative risk assessment for all of these chemicals is not practical. Consequently, a set of chemicals that represents the greatest public health risk has been chosen for modeling and risk assessment. The methodology for selecting these indicator chemicals is described in the Superfund Public Health Evaluation Manual (SPHEM) (EPA 1986) and is discussed briefly below. The method of selection of indicator chemicals takes into account a substance's toxicity, concentration, mobility, and persistence in the environment.

## 5.2 INDICATOR SCORES

An indicator score is a measure of a contaminant's potential impact on human health. It is the product of a representative contaminant concentration and a toxicity factor and is calculated for each compound or element present in significant concentrations for each potential migration pathway (i.e., water, air, or soil). Chemical toxicity and carcinogenicity are considered separately. Carcinogens, therefore, may have two toxicity constants and, correspondingly, two indicator scores: a score for chemical toxicity and a score for cancer effects. Indicator scores for the different types of effects are not directly comparable. A separate total indicator score for each substance for each type of effect is obtained by adding the appropriate indicator scores for each pathway.

Indicator scores calculated for water and air pathways for the VOCs (which are the chief contaminants at the Oil Landfarm) are shown in Table 5.1; scores for the soil/sediment pathway are negligible for all contaminants. Indicator scores cannot be calculated for the PCBs because no toxicity factors are available. True air-pathway indicator scores cannot be calculated because there are currently no data available on concentrations in air. The air-pathway indicator scores shown in Table 5.1 are calculated from groundwater concentrations; this approach assumes that air concentrations are related to groundwater concentrations by a proportionality factor which is the same for all contaminants. This assumption is probably valid for the main potential source of air contamination, the off-gas from the treatment modules (see Sect. 6). Air-pathway indicator scores obtained this way can be compared to each other. However, they cannot be added to water-pathway scores to give total scores; therefore, water- and air-pathways are treated separately in Table 5.1. As this table illustrates, in both water and air, TCE, PCE, and 1,1-DCE are the compounds with high indicator scores for both chemical toxicity and carcinogenicity.

## 5.3 OTHER FACTORS

Other factors to be considered when choosing indicator chemicals are mobility and persistence in environmental media. If a contaminant is much more mobile or persistent than other chemicals present, it should be included among the indicator chemicals regardless of its

indicator score. The transport media of most concern at the Oil Landfarm WMA are groundwater and surface water. The mobility of a chemical in water depends on its solubility and its propensity for adsorption onto solids. These properties are discussed in detail in Sect. 7.2; we simply note here that the VOCs with the highest indicator scores include those substances which can be expected to be most mobile. PCBs are highly immobile in aqueous systems.

**Table 5.1. Indicator scores**

| Contaminant | Indicator score[a] | | | | Selected as indicator chemical |
| | Chemical toxicity | | Cancer effects | | |
| | Water | Air | Water | Air | |
| --- | --- | --- | --- | --- | --- |
| Trichloroethene | 0.44 | 12.4 | 0.002 | 0.02 | XX |
| 1,1-Dichloroethene | 0.23 | 3.5 | 0.08 | 0.76 | XX |
| Tetrachloroethene | 0.004 | 0.01 | 0.003 | 0.04 | XX |
| trans-1,2-Dichloroethene | 0.01 | 0.08 | b | b | |
| Vinyl chloride | 0.002 | 0.02 | 0.000 | 0.001 | |
| 1,1-Dichloroethane | 0.05 | 0.5 | b | b | |
| Benzene | 0.004 | 0.000 | 3.8 | 0.002 | |
| 1,1,1-Trichloroethane | 0.000 | 0.003 | b | b | |
| Polychlorinated biphenyls[c] | | | | | XX |

[a]Product of the average measured 1988 concentration (in the well in which the highest concentration of the contaminant occurred) and toxicity factor (EPA 1986).
[b]No toxicity factor given.
[c]Contaminant only of soils and sediments (see text).

Organic compounds may undergo transformation processes such as biodegradation, hydrolysis, or photolysis. However, as discussed in more detail in Sect. 7.2.4, all the VOCs and PCBs tend to be quite persistent in the environment, and there is currently not enough information available to rank them according to relative persistence.

## 5.4 SELECTION OF INDICATOR CHEMICALS

The indicator chemicals selected for this study are PCE, TCE, 1,1-DCE, and PCBs. The selection of the VOCs is based primarily on indicator scores. PCBs are included because of their relatively high concentrations in soils and sediments and their high cancer potency factor (CPF).

# 6. PROPOSED REMEDIAL ALTERNATIVES

## 6.1 CAPPING

In the last year large volumes of contaminated soils have been excavated from the Oil Landfarm disposal plots, and the plots and the Chemical Storage Area have been covered with engineered caps. Capping of the Boneyard area is under way. The caps consist of integrated systems of impermeable and drainage layers, which prevent infiltration of surface water into the wastes. The completed caps will cover approximately the shaded area in Fig. 6.1. A single cap covers the three Oil Landfarm disposal units, and the Boneyard cap will join the cap over the Chemical Storage Area (Martin Marietta Energy Systems, Inc. 1988; Lockwood Greene Engineers, Inc. 1988). The Sanitary Landfill was capped with clay and revegetated in 1983.

## 6.2 CAPPING AND GROUNDWATER RECOVERY AND TREATMENT

In addition to the construction of the cap, under this option contaminated groundwater would be extracted by a recovery well network, and the recovered water would be treated in a facility to be built at the Bear Creek Burial Grounds along with contaminated water from that site. Completion of the well network at the Oil Landfarm WMA is scheduled for 1993 (Martin Marietta Energy Systems, Inc. 1988). No engineering design of the recovery well network is currently available. A conceptual well network was designed as part of this study. It consists of six wells, each pumping at 2 gal per minute (gpm), and is described further in Sect. 7.3.3.4. The base-case design for the groundwater treatment facility includes air stripping and activated carbon filtration, although other technologies will be evaluated further before a final system is selected. Currently, there are no plans to treat the air exiting the facility, and air stripping will simply transfer volatile contaminants to the air. The point of discharge of treated water is undecided; two possibilities are East Fork Poplar Creek and Bear Creek.

## 6.3 COSTS OF THE ALTERNATIVES

Cost estimates for the remedial alternatives for the Oil Landfarm WMA were performed during preparation of the Closure and Postclosure Activities Remedial Action Program Description document (Martin Marietta Energy Systems, Inc. 1988). The Bear Creek Burial Grounds groundwater treatment facility was then conceived of as a dual-phased facility, with the first phase treating 50 gpm of contaminated groundwater from both the Burial Grounds and the Oil Landfarm areas, and the second phase increasing the overall treatment rate to 114 gpm. The total capital cost estimated for Phase I is $10.6 million; for Phase II, $8.6 million. Operating and maintenance costs (not including utility or byproduct disposal costs) are estimated at $2 million/year for the postclosure treatment facilities. Estimated total costs for capping the Oil Landfarm WMA (not including the Sanitary Landfill) are $8 million (Harper, personal communication). Operating and maintenance costs for the cap are expected to be minimal.
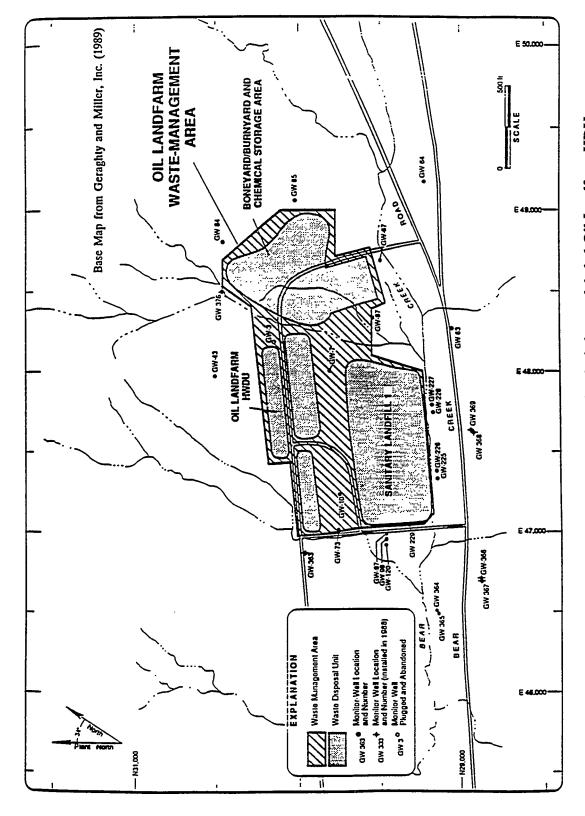
Fig. 6.1. Engineered cap will cover approximately the shaded areas labeled Oil Landfarm HDU, Boneyard/Burnyard, and Chemical Storage Area. A clay cap currently covers the area labeled Sanitary Landfill 1.

In preparing risk assessments for both the Burial Grounds and the Oil Landfarm WMA, more realistic groundwater collection networks were analyzed. The above cost estimates are not directly applicable to these networks. However, we expect costs associated with the recovery networks evaluated in this report to be comparable.

# 7. ENVIRONMENTAL TRANSPORT AND PERSISTENCE OF CONTAMINANTS

## 7.1 INTRODUCTION

As discussed in Sect. 4, groundwater is the most seriously contaminated medium at the Oil Landfarm WMA, and VOCs are the chief contaminants of groundwater. Currently, groundwater contamination does not extend beyond a few hundred feet from the perimeter of the waste management area. Uranium is the only contaminant present at significant levels in surface water and sediments in the vicinity of the site. Soils within the waste management area contain elevated levels of VOCs, PCBs, and uranium.

Although, at present, contamination is confined to the immediate vicinity of the waste management area, with time contaminants may migrate farther from their sources in the disposal areas. Potential migration pathways are: transport via groundwater, surface water, and air, and movement with soils and sediments. The extent to which contaminants migrate in these media depends on both the properties of the contaminants and the properties of the media. At the Oil Landfarm WMA, the media with the greatest potential for transporting contaminants are groundwater, surface water, and air.

In this study, computer modeling is used to predict the future migration of the indicator chemicals in groundwater under the three remedial alternatives. Initial air concentrations for the VOCs, which would be emitted from the groundwater treatment facility, are also estimated from computer modeling, using input from the groundwater modeling. Contaminant migration via surface water, soil, and sediment movement are treated qualitatively. Groundwater modeling is discussed in detail in Appendix A, and air modeling is discussed in this section. The results of all these efforts and some relevant assumptions are discussed in the following subsections.

## 7.2 CONTAMINANT PROPERTIES

The behavior of contaminants in the environment, especially their mobility, is largely determined by their physical and chemical properties. The most important of these properties are solubility, propensity for adsorption, volatility, and susceptibility to transformation processes. The following sections discuss these properties for the contaminants of concern and how these properties relate to conditions at the site.

### 7.2.1 Solubility

Solubility values for the contaminants of concern at the Oil Landfarm WMA are shown in Table 7.1. All the VOCs are only slightly to moderately water soluble (solubilities ranging from 150 to 2250 mg/L). Low solubilities, however, do not prevent the VOCs from being transported as dissolved phases in groundwater, as is evident from water quality data for the site. PCBs are relatively insoluble in water and are, therefore, not readily transported via water pathways.

Table 7.1. Contaminant properties[a]

| Contaminant | Solubility[b] (mg/L) | log $K_{ow}$[c] | log $K_{ow}$[b] | Density[c] (g/cm$^3$) | Henry's law[b] constant (atm-m$^3$/mol) |
|---|---|---|---|---|---|
| Tetrachloroethene | $1.50 \times 10^2$ | 3.14 | 2.60 | 1.62 | $2.59 \times 10^{-2}$ |
| Trichloroethene | $1.10 \times 10^3$ | 2.42 | 2.38 | 1.47 | $9.10 \times 10^{-3}$ |
| 1,1-Dichloroethene | $2.25 \times 10^3$ | 2.13 | 1.84 | 1.21 | $3.40 \times 10^{-2}$ |
| Polychlorinated biphenyls | $3.10 \times 10^{-2}$ | | 6.04 | 1.44 | $1.07 \times 10^{-3}$ |

[a]Solubility, log $K_{ow}$, and Henry's law constant for neutral pH and 20 to 30°C.
[b]Source: EPA 1986.
[c]Source: Arthur D. Little, Inc. 1985.

Because of their low solubilities, some portion of the VOCs deposited in the Oil Landfarm WMA almost certainly exist as nonaqueous phase liquids (NAPLs) rather than as dissolved constituents of groundwater or adsorbates. Because the VOCs are also much denser than water (Table 7.1), there is a strong possibility that they have penetrated the saturated zone as NAPLs. Experimental evidence indicates that the movement of dense NAPLs in the saturated zone is little affected by the regional hydraulic gradient, and that they can be expected to move essentially vertically downward (Schwille 1988; Anderson, Johnson, and Pankow 1987). Their behavior in fractured media and porous media differ in detail, but the overall result is much the same: over time they can be expected to persist in a column below the original source; this column consists of NAPL droplets entrained in pore-space water or fractures and pools of NAPL formed above impermeable strata. Figure 7.1 illustrates this situation.

Dipping bedding planes, joint sets, and solution cavities at the Oil Landfarm WMA site would further complicate this picture. The column itself is relatively immobile in the horizontal plane, and groundwater contamination away from the source area results from the dissolution of the NAPLs by groundwater passing through the column. The NAPL column acts essentially as a long term source of contaminants to the aquifer. Hunt et al. (1988), on the basis of analytical estimates, predict that the time for NAPL dissolution into groundwater is on the order of decades. Possible implications of this behavior for this study are discussed in later sections.

## 7.2.2 Sorption

Another chemical property influencing contaminant migration and persistence is the contaminant's potential for sorption to the surrounding aquifer matrix. The octanol-water partition coefficient ($K_{ow}$) can be used to estimate an organic compound's sorption potential or its distribution coefficient (Table 7.1). Octanol-water partition coefficients found in the literature vary, but in general they indicate that the VOCs have moderate potentials for
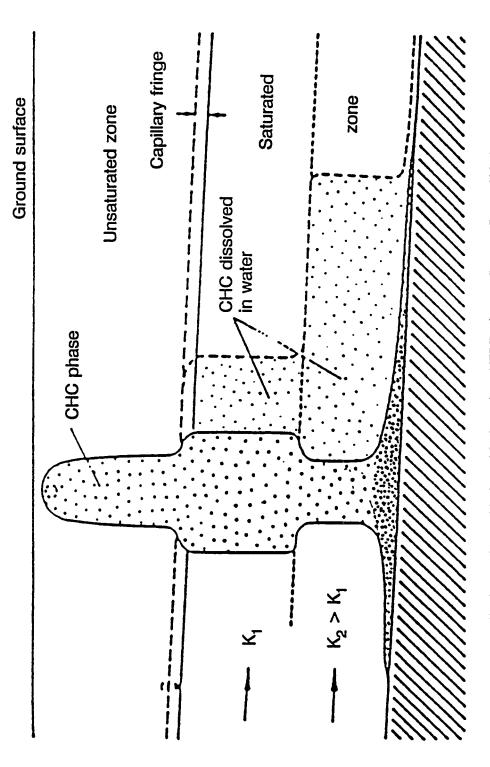
Fig. 7.1. Distribution of dense chlorinated hydrocarbon (CHC) phases. *Source: Dense Chlorinated Solvents in Porous and Fractured Media,* by Friedrich Schwille, translated by James F. Pankow. Copyright 1988, Lewis Publishers, Inc, Chelsea, Mich. Used with permission.

adsorption, and PCBs have very large potentials for adsorption. For this study the most important effect of adsorption is to slow the transport of the VOCs in groundwater. Modeling this effect is discussed in detail in Appendix A. Of the contaminants modeled, PCE has the lowest transport velocity and 1,1-DCE has the highest.

### 7.2.3 Volatility

A contaminant's tendency to volatilize is also an important factor in its migration and persistence. As indicated by the Henry's law constants listed in Table 7.1, the VOCs are highly volatile from water. Volatilization is thus an important loss mechanism for VOCs from surface waters and near surface soils. In addition, high volatility makes air stripping a feasible remedial alternative. Although PCBs have appreciable volatility from water, volatilization is generally reduced by their sorption to particles.

### 7.2.4 Transformation Processes

Finally, contaminant migration and persistence can be greatly influenced by abiotic and biotic transformation processes. Organic compounds are particularly susceptible to these changes. The occurrence and distribution of VOCs at the Oil Landfarm WMA indicate that transformation processes are taking place at the site. Two major associations of contaminants are observed: PCE + TCE + trans-1,2-dichloroethene and 1,1,1-trichloroethane + 1,1-dichloroethane + 1,1-DCE (Table 4.3). The first association is probably related to the following biodegradation chain (Wood, Lang, and Payan 1985; Parsons, Wood, and DeMarco 1984):

$$\text{PCE} \rightarrow \text{TCE} \rightarrow \left\{ \begin{array}{l} \textit{cis}\text{-1,2-dichloroethene} \\ \textit{trans}\text{-1,2-dichloroethene} \\ \text{1,1-dichloroethene} \end{array} \right\} \rightarrow \text{vinyl chloride} .$$

The second association can be explained by the transformation of 1,1,1-trichloroethane abiotically to 1,1-DCE and biotically to 1,1-dichloroethane as documented by Vogel and McCarty (1987). Unfortunately, not enough data are available to treat these processes quantitatively. The continued presence of PCE, TCE, and 1,1,1-trichloroethane at the site suggests that the half lives of these compounds must be relatively long, and the continued presence of the dichloroethenes and of 1,1-dichloroethane indicates that their subsequent transformations to vinyl chloride and chloroethane, respectively, must also have long half lives. Consequently, in this study all VOCs are treated as independent entities.

## 7.3 CONTAMINANT MIGRATION AND PERSISTENCE IN GROUNDWATER

### 7.3.1 Description of the Groundwater Model

Contaminant transport in groundwater is simulated using the U.S. Geological Survey (USGS) Two-Dimensional Solute Transport Method of Characteristics Model (MOC) (Konikow and Bredehoeft 1978). The model solves two simultaneous partial differential equations: the groundwater flow equation, which describes the distribution of hydraulic heads in the aquifer, and the solute transport equation, which gives the distribution of solute

concentrations in the aquifer. The model is based on a rectangular, block-centered, finite-difference grid which corresponds to the surface area of the aquifer. In this study flow is treated as steady-state. In addition to convective transport, hydrodynamic dispersion and adsorption are simulated. Transformation processes are not treated. Several aquifer and contaminant properties must be estimated and input to the model, and the model calibrated by matching current hydraulic head and contaminant distributions by model output. These aspects of modeling are discussed in detail in Appendix A. The model output includes maps of hydraulic heads and concentrations of dissolved contaminants at finite-difference nodes at given times and concentration vs time data at selected locations.

## 7.3.2 Modeling Assumptions

Three assumptions germane to the Oil Landfarm WMA simulations are inherent in the MOC model: (1) the aquifer is in a granular porous medium, (2) aquifer properties and contaminant distributions are uniform with depth, and (3) contaminants are transported as a single aqueous phase. The reasons that each of these assumptions may be questionable for the Oil Landfarm WMA are discussed below. Ideally, making these assumptions could be avoided by using more complicated models; however, in practice there are currently insufficient data to justify doing so for the purpose of risk assessment. The model limitations do not seriously affect the usefulness of the results; they should, however, be kept in mind when interpreting them.

The first assumption, that the permeability of the aquifer is produced by intergranular porosity, is questionable for the Oil Landfarm WMA because, as discussed in Sect. 2, in most of the formations of the Conasauga Group the permeability is created chiefly by fractures and, in the case of the Maynardville Limestone, partly by cavities. Although using porous media models for fractured media is common practice, there are potentially significant differences in the behavior of groundwater flow and contaminant transport for the two types of media (Huyakorn, White, and Wadsworth 1987; Sudicky and Frind 1984). These differences are difficult to assess qualitatively, and there is at present no consensus on the conditions for which the use of porous media models for transport in fractured media is justified. Results of aquifer tests conducted in the Nolichucky in Bear Creek Valley (Lozier, Spiers, and Pearson 1987) do, however, suggest that the steady-state response of the aquifer can be adequately modeled with a porous media model such as MOC. Although some fractured media effects were observed in these tests, drawdown curves for most wells corresponded to the classic porous media pattern.

Use of a porous media model for a cavity-ridden formation such as the Maynardville involves more uncertainty. Hydraulic and physical properties, and, consequently, flow paths and contaminant transport in such media can be quite erratic. Fortunately, because of the geology and topography of the site, flow paths are likely to be constrained to lie along the trend of the valley floor, giving confidence in the overall transport direction. However, the scale of this study is probably not large enough that hydraulic and physical properties such as transmissivity and porosity (which determine transport velocities) can be adequately represented by averages. Consequently, one should not expect the simulations to accurately predict detailed distributions of contaminants in the Maynardville, but only to give a general sense of the extent of plume migration.

The second potential limitation is that the model is two-dimensional. This implies that aquifer properties and concentrations are uniform with depth and, consequently, that groundwater flow and contaminant transport are entirely horizontal. Aquifer properties at the Oil Landfarm WMA are not uniform with depth, nor are contaminant concentrations, as indicated by monitoring well data. In addition, monitoring well data and preliminary modeling (Bailey 1988) indicate vertical components to groundwater flow, especially in the vicinity of Bear Creek. However, given other uncertainties in the simulations and the paucity of data on the vertical components of flow, a two-dimensional treatment with aquifer properties averaged or integrated over the vertical dimension (see Appendix A) appears to be adequate for the purpose of risk assessment.

The third assumption in the model of potential significance to the simulations is that contaminants in the aquifer are transported in the aqueous phase. This assumption is related to the problem of vertical flow but is treated separately here because it has additional ramifications. As discussed above, the VOCs are only slightly water soluble and are much denser than water; therefore, large volumes of these compounds have probably entered the aquifer as nonaqueous phase liquids (NAPLs). The MOC model clearly cannot describe contaminant transport in a NAPL column. However, the column itself is relatively immobile in the horizontal plane, and groundwater contamination away from the source area can be adequately modeled by MOC. The consequences of the probable long-term NAPL sources are discussed below.

### 7.3.3 Modeling Remedial Alternatives

#### 7.3.3.1 Introduction

Three possible remedial alternatives for the Oil Landfarm WMA are modeled: (1) take no action; (2) place an impermeable cap over the disposal areas of the Oil Landfarm WMA; and (3) in addition to the cap, construct a network of recovery wells to extract contaminated groundwater for treatment. Two cases are considered for the recovery well option: wells pumping over only the upper 200 ft of the aquifer and wells pumping over 600 ft of the aquifer. In addition, we attempt to estimate the effects of long-term NAPL sources on the capping and recovery well options. For each alternative, model simulations are run for 100 years from 1988; the initial conditions are the calibrated 1988 simulation results (see Appendix A). The engineering parameters for the cap and construction schedules are based on the Remedial Action Program Description (Martin Marietta Energy Systems, Inc. 1988). The recovery well network was designed as part of the present study. This design is entirely conceptual and is intended only to suggest whether or not groundwater recovery may be a productive option to pursue.

#### 7.3.3.2 No action

The locations of model contaminant sources are based on the historical data described in Sect. 3, geophysical data and current soil and groundwater contamination patterns (see Appendix A). These data indicate that there are three major source areas associated with the site. In terms of the contaminants of concern, there are two primary PCE and TCE sources: one located in the vicinity of the southwest corner of the Boneyard/Burnyard or the Chemical Storage Area and the other near the southeast group of Oil Landfarm plots. The source of 1,1-DCE appears to be located near GW-3 and was probably originally associated with the

northern Oil Landfarm plots. Source terms in the simulations are assigned to the finite-difference cells approximately corresponding to these locations (see Appendix A). We did not attempt to simulate the TCE contamination along the trend of Bear Creek in the Maynardville, but it appears to originate at a source upstream from the Oil Landfarm WMA.

Pre-1988 source-term parameters are poorly constrained, and the current spatial distribution of groundwater contamination introduces considerable uncertainty into the calibration of the transport simulations (see Appendix A). Pre-1988 source terms are based almost entirely on best fits to concentration data at the locations of GW-87, GW-7, and GW-3, and calibration is judged primarily by the goodness of fit, i.e., how well the simulation matches concentrations at these wells. Consequently, the applicability of simulated 1988 concentrations (which are the input to all other runs) to much of the area of Oil Landfarm WMA is uncertain. Ideally, post-1988 source terms should decrease with time because contaminants are no longer being introduced into the WMA. There are, however, no data from which to estimate the form and the rate of decrease of source terms. As a worst-case scenario, therefore, this study uses post-1988 source terms which are constant and of the same magnitude as pre-1988 calibrated source terms. This approach almost certainly overestimates post-1988 simulated no-action concentrations, but does not imply altogether unreasonable amounts of contaminants; for example, PCE source terms running for 115 years input less than six 55-gal drums of the compound to the aquifer. Because of the large uncertainty in source terms, predicted concentrations for the no-action (and NAPL-source scenarios) should be considered within an order of magnitude.

### 7.3.3.3 Capping

The final caps cover all disposal areas of the Oil Landfarm WMA except the Sanitary Landfill (Fig. 6.1). Construction of the caps began in 1989 and is scheduled to be completed in mid-1991. Simulated no-action concentrations for 1988 and 1991 are not significantly different. For convenience, therefore, the simulations assume cap emplacement in 1988. Two opposite capping scenarios are considered: (1) a fully effective cap, which eliminates the sources of contaminants to the aquifer and (2) a completely ineffective cap, which does nothing to decrease contaminant input to the aquifer because the contaminants are already in the saturated zone as NAPLs. A fully effective cap is represented in the simulations by making two modifications to the base model. First, because the cap is impermeable, annual recharge in the cells corresponding to the capped area is set to zero after 1988. The second, and more important, change is that after 1988 the source terms are also set to zero. The rationale for setting the source terms to zero is that the impermeable cap prevents leachate production by percolation of rainwater through the wastes. However, large quantities of VOCs may already be in the saturated zone as NAPLs. The dissolution of these compounds into the groundwater would not be prevented by the presence of a cap, and they would, therefore, act as continuing sources of contamination to the aquifer. This situation constitutes the completely ineffective cap scenario, and modeling the scenario requires long-running source terms in the simulations. Using pre-1988 source terms, which run throughout the 100 years of simulations (see above), in the capping simulations produce results that are virtually indistinguishable from the results for the no-action alternative. Simulations with no-source terms represent the best possible results obtainable by capping, and simulations with source terms can be expected to resemble worst-case results; reality is likely to lie somewhere between the two cases.

## 7.3.3.4 Cap and recovery wells

In addition to the modifications to the base model which are made to represent the cap, this option requires the addition of several extraction wells to the model. The recovery well network is scheduled for completion in 1993, and recovery well simulations begin then using initial concentrations predicted from capping simulations. Again, two scenarios are considered: one in which the cap is assumed to be completely effective and source terms are zero after 1988 and one in which the cap is considered to be completely ineffective and source terms run throughout the simulations. The well network design is the same in either case and is based on the capture-zone technique described by Javandel and Tsang (1986). The basic idea is that pumping wells located at the downgradient edge of the plume alter the hydraulic head distribution so that all contaminated groundwater within a certain region or capture zone flows to them and can be extracted (Fig. 7.2). The size of the capture zone is proportional to the pumping rate and inversely proportional to the aquifer depth and the Darcy velocity. Geraghty and Miller (1987f) estimate that a reasonable pumping rate per well for the Oil Landfarm WMA aquifer is approximately 2 gpm. Because of this low pumping rate, several spatially separated extraction wells are necessary to create a capture zone large enough to significantly impede plume migration.

The location and geology of the site restrict the placement of recovery wells. To effectively contain all of the predicted 1993 plume it would be necessary to locate wells in the Maynardville Limestone along the trace of Bear Creek. Locating wells in that area is problematic for two reasons. First the hydraulic behavior of wells in a cavity-dominated formation such as the Maynardville can be expected to be quite erratic, and is virtually impossible to predict using computer simulations of the type done in this study. Second, extraction wells located near Bear Creek would derive a large fraction of their discharges from induced infiltration from Bear Creek and the underlying cavity system (Newsom and Wilson 1988) and would, therefore, not be efficient in removing contaminants from the Oil Landfarm WMA site. A further limitation on the placement of wells is the existence of the Sanitary Landfill over part of the predicted plume area. There are potential problems with drilling wells through the landfill; however, simulations with wells in this area were run to check whether it might be a useful option to consider. These simulations showed only slightly more efficient recovery of contaminants than did the final model recovery well network. The final well network is shown in Fig. 7.3. For these reasons the recovery well network does not contain wells located in the Maynardville or in the area of the Sanitary Landfill.

The depth to which significant contamination occurs in the aquifer at the Oil Landfarm WMA is not well characterized. However, water quality data indicate that serious contamination extends to at least 200 ft in the Nolichucky (the deepest well at the site is screened at 245 ft). In addition data from the Bear Creek Burial Grounds site, located less than a mile away down Bear Creek Valley, indicate high VOC concentrations in a well over 500 ft deep in this formation. Ideally, a recovery well network might be designed to extract contaminants over this depth. However, measured hydraulic conductivities in the vicinity of 500 ft are so low that it may be impractical to try to recover contaminants located there. Accordingly, two limiting recovery well cases are modeled. In the first the extraction of contaminated water over 600 ft of aquifer is simulated; in the second, extraction of contaminated water over only the upper 200 ft of aquifer is simulated. The configuration of the optimum recovery well network is the same for both depths.

## FOUR-WELL CAPTURE-ZONE TYPE CURVES
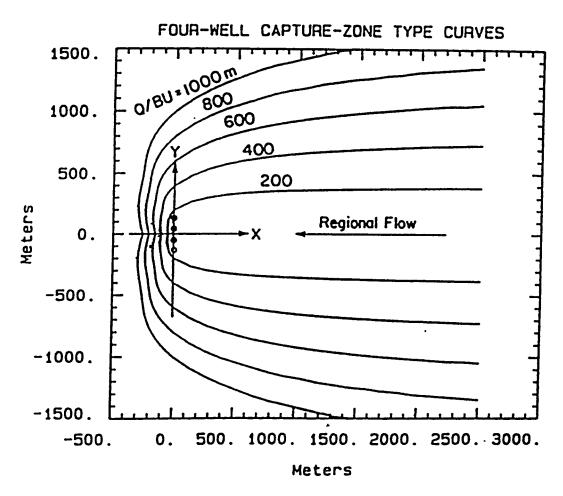


Fig. 7.2. Type curves showing capture zones of four wells for several values of Q/BU, where Q is the pumping rate, B is the aquifer depth, and U is the regional Darcy velocity. *Source:* I. Javandel and C. Tsang 1986. "Capture-Zone Type Curves: A Tool for Aquifer Cleanup," Ground Water, 24(5), 616–626. Reprinted with permission from Anita B. Stanley. National Water Well Association, Dublin, Ohio.
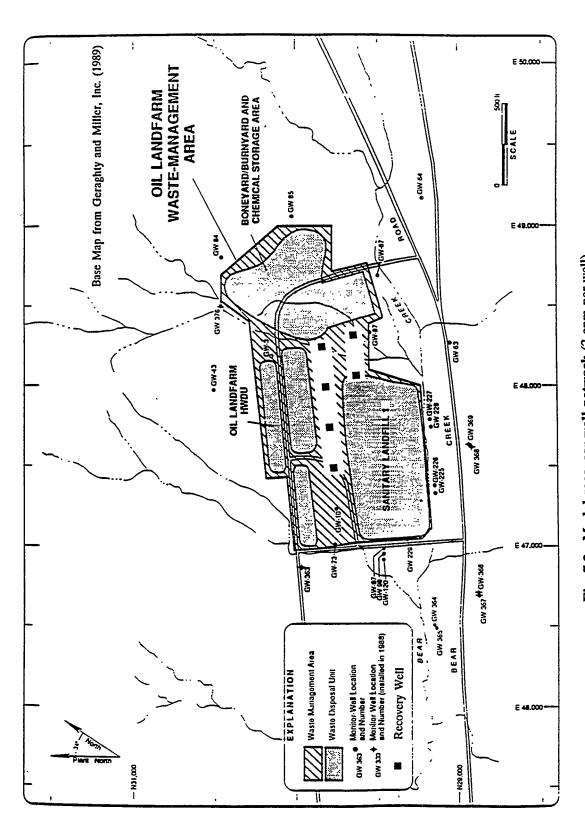
Fig. 7.3. Model recovery well network (2 gpm per well).

### 7.3.4 Model Predictions of Contaminant Transport and Persistence

As discussed above, it is not possible to predict the actual effectiveness of the caps, and we have, consequently chosen to model two limiting scenarios: capping which eliminates sources of contaminants to the aquifer and capping which does not affect sources of contaminants. The addition of these scenarios to the three alternatives and the two recovery well depths necessitates considering a large number of cases and complicates the presentation of results. The primary purpose of this analysis, however, is to compare the probable effectiveness of capping alone with the effectiveness of capping plus groundwater recovery. To focus on this comparison we first present capping and capping plus recovery results assuming an effective cap and then present the results for these alternatives assuming an ineffective cap. As will be demonstrated, the predicted relative efficacy of alternatives does not depend on assumptions about whether or not the cap eliminates sources of contaminants.

#### 7.3.4.1 No action

Figures 7.4–7.12 illustrate the predicted concentrations of the indicator VOCs for the no-action alternative. These figures show concentration contours at three times: 15, 30 and 60 years from 1988. In these contour plots the 10 $\mu g/L$ isopleth is used to represent the plume fronts because it is above numerical noise in the simulations and is, therefore, a reliable indicator of transport distance. The figures show that the transport direction is south to southwest as would be expected from both the measured and the modeled water table configurations and the anisotropy of the aquifer (see Figs. 2.4 and A.4).

Even with continuous source terms, contaminants migrate less than 2000 ft from the southwest corner of the WMA in groundwater in the 100 years of the simulations. There are two primary factors limiting migration of the contaminant plumes. First, the fronts of the plumes rather quickly reach a model region where reversed hydraulic gradients (which represent increasing topographical elevations on Chestnut Ridge) arrest further migration to the south. Second, contaminants at the fronts of the modeled plumes intersect sinks in the model (which represent groundwater discharge into Bear Creek and the shallow cavity system) and are removed from the modeled groundwater system. The plumes reach a steady-state condition in which contaminant input from the source terms is balanced by contaminant discharge to the Bear Creek sinks; the greater the contaminant's transport velocity, the more quickly it reaches steady state.

For the no-action alternative the concentrations of PCE, TCE, and 1,1-DCE remain at levels greater than 500 $\mu g/L$ beneath the area of the WMA and at levels of 10 to 200 $\mu g/L$ along the floor of Bear Creek Valley for the entire 100 years of simulation. The predicted persistence of these compounds is due to the constant source terms, which run throughout the simulations. The predicted concentrations in the plume cores strongly depend on the source terms and should, therefore, be considered order of magnitude estimates only.

#### 7.3.4.2 Capping without source terms

Figures 7.13–7.18 show the results of the capping simulations without source terms at 15 and 30 years from 1988. Comparison of these figures with the no-action results at corresponding times shows that, although the plume fronts are in about the same locations, the areas and the peak concentrations in the plumes produced by the capping without
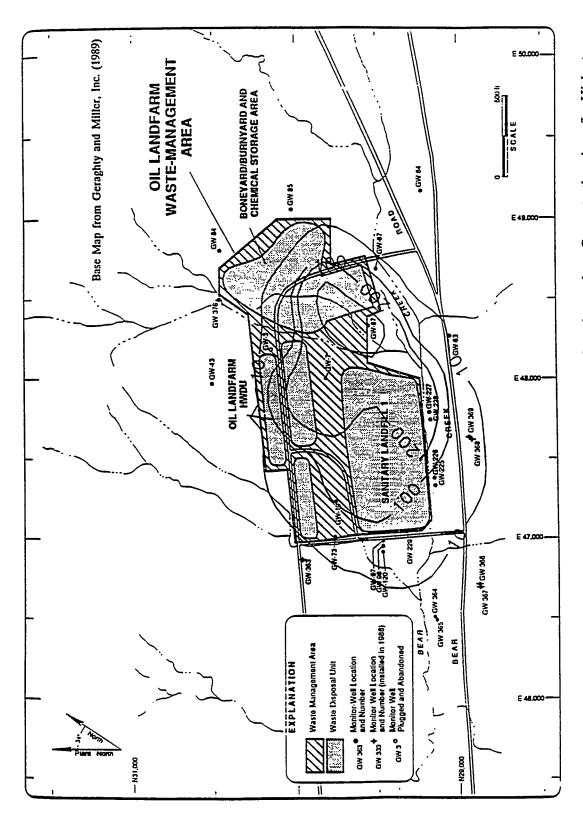
Fig. 7.4. PCE distribution at 15 years from 1988 for the no-action alternative. Concentrations in $\mu g/L$. Highest contour 500 $\mu g/L$.
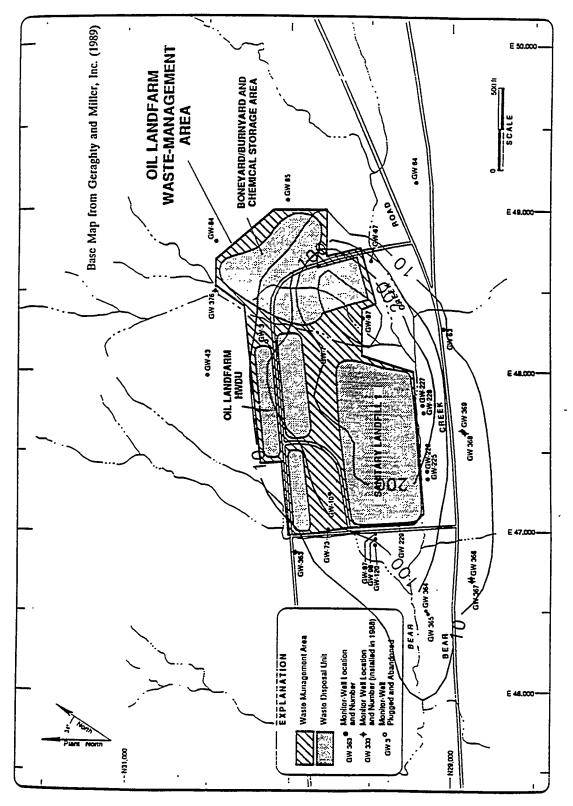
Fig. 7.5. PCE distribution at 30 years from 1988 for the no-action alternative. Concentrations in μg/L. Highest contour 500 μg/L.
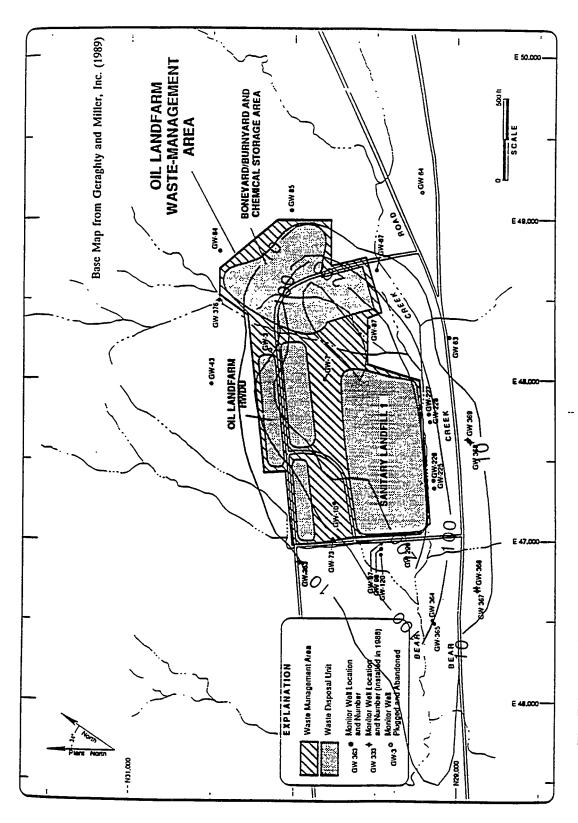
Fig. 7.6. PCE distribution at 60 years from 1988 for the no-action alternative. Concentrations in μg/L. Highest contour 500 μg/L.
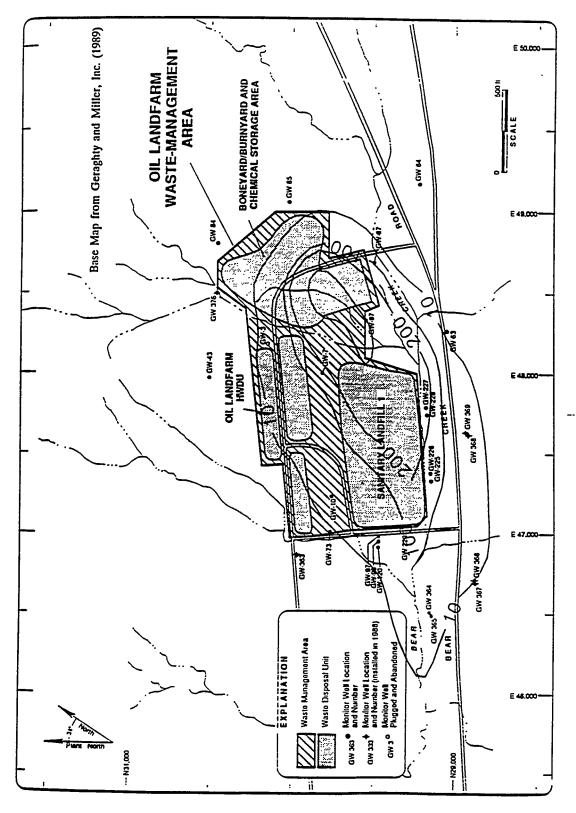
Fig. 7.7. TCE distribution at 15 years from 1988 for the no-action alternative. Concentrations in $\mu g/L$. Highest contour 500 $\mu g/L$.
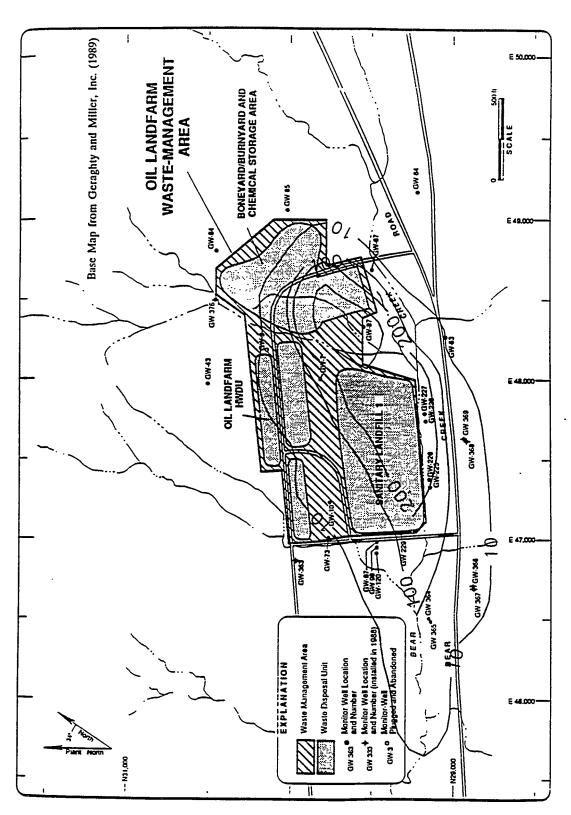
Fig. 7.8. TCE distribution at 30 years from 1988 for the no-action alternative. Concentrations in $\mu g/L$. Highest contour 500 $\mu g/L$.
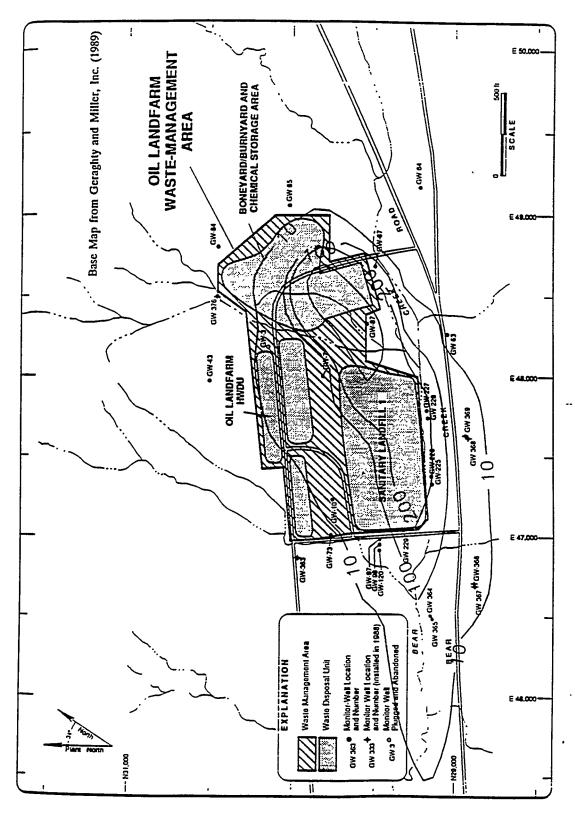
Fig. 7.9. TCE distribution at 60 years from 1988 for the no-action alternative. Concentrations in $\mu g/L$. Highest contour 500 $\mu g/L$.

48
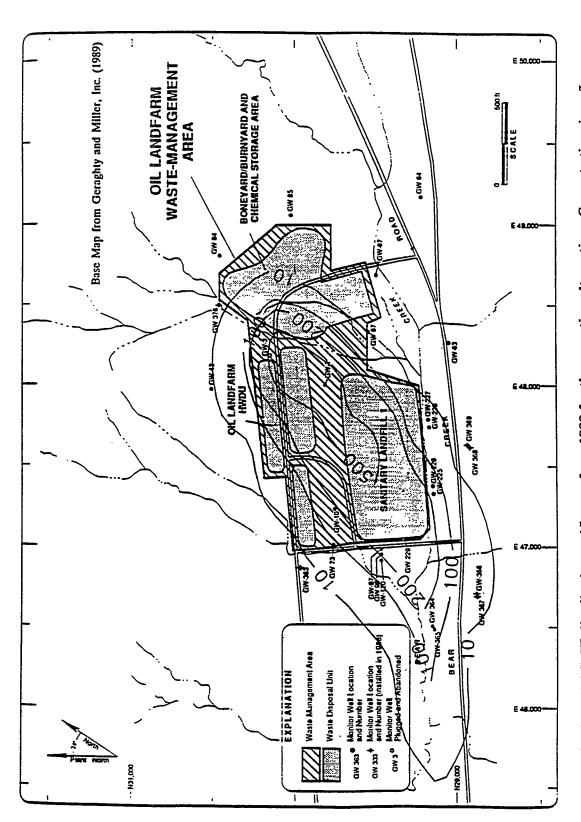


Fig. 7.10. 1,1-DCE distribution at 15 years from 1988 for the no-action alternative. Concentrations in $\mu g/L$.
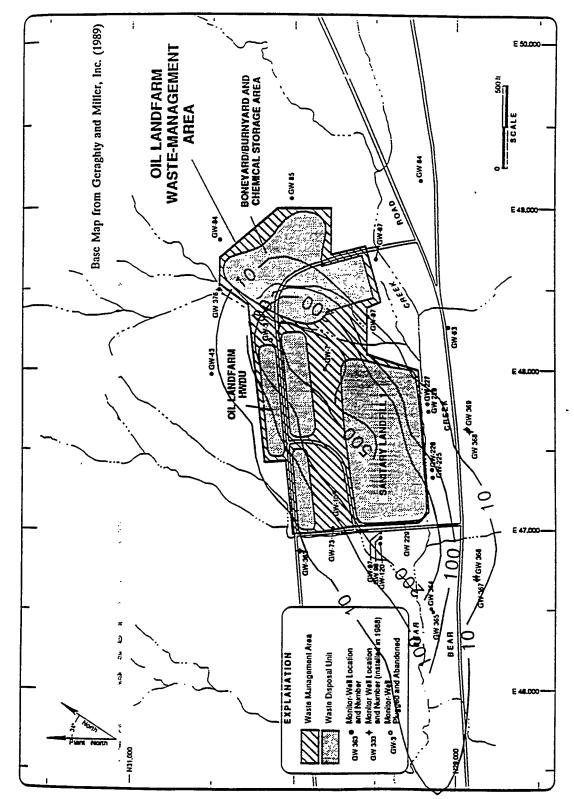
Fig. 7.11. 1,1-DCE distribution at 30 years from 1988 for the no-action alternative. Concentrations in $\mu g/L$.
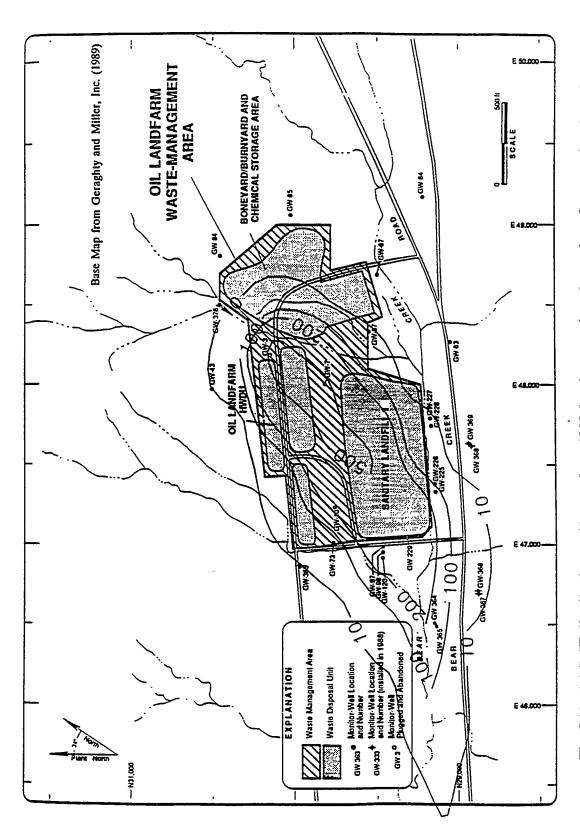
Fig. 7.12. 1,1-DCE distribution at 60 years from 1988 for the no-action alternative. Concentrations in $\mu g/L$.

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Fig. 7.13. PCE distribution at 15 years from 1988 for the capping alternative without source terms. Concentrations in μg/L. Highest contour 200 μg/L.

Fig. 7.14. PCE distribution at 30 years from 1988 for the capping alternative without source terms. Concentrations in μg/L.

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**Fig. 7.15. TCE distribution at 15 years from 1988 for the capping alternative without source terms. Concentrations in μg/L. Highest contour 200 μg/L.**

Fig. 7.16. TCE distribution at 30 years from 1988 for the capping alternative without source terms. Concentrations in μg/L.

Fig. 7.17. 1,1-DCE distribution at 15 years from 1988 for the capping alternative without source terms. Concentrations in μg/L.

Fig. 7.18. 1,1-DCE distribution at 30 years from 1988 for the capping alternative without source terms. Concentrations in $\mu g/L$.

source-term simulations are significantly smaller than those from the no-action simulations. In addition, the capping plumes persist for much shorter times: the 1,1-DCE plume is almost gone (i.e., concentrations are about 10 $\mu$g/L) by 30 years; the TCE plume completely dissipates by about 45 years; and the PCE plume is gone by about 60 years. These effects occur because the mass in the plumes is not continuously replenished by source terms in the simulations, and all contaminants eventually discharge to the Bear Creek sinks. The cores of plumes of contaminants with higher transport velocities reach these sinks earlier and dissipate more quickly. As discussed above, the results presented in Figures 7.13–7.18 are likely to be overly optimistic, and the actual situation at a given time may more resemble the no-action results.

### 7.3.4.3 Capping and recovery wells without source terms

Figures 7.19–7.24 show the results of the 600-ft recovery simulations without source terms, and Figs. 7.25–7.29 show the results for the 200-ft case, also without source terms. The simulations predict that recovery in the 600-ft case is little more effective than capping in reducing contaminant concentrations, and that cleanup times are about the same for these two alternatives. The 200-ft recovery case is somewhat more efficient. For this case peak concentrations and plume areas are both reduced by about a factor of 2 with respect to the capping and 600-ft results at given times, and overall cleanup times are 5 to 10 years earlier. The simulations in which the recovery wells pump over only 200 ft produce a larger capture zone and show faster reductions in plume concentrations because only one third as much mass must be removed over a depth of 200 ft as over a depth of 600 ft to obtain an equivalent decrease in concentration.

The recovery well networks do not capture all of the plumes partly because of the low pumping rates used and partly due to the restrictions on the locations of wells; the capture zones created are not large enough to contain the plumes. However, predicted 1988 concentrations, upon which the initial concentrations to the recovery runs depend, are quite uncertain under the area to the south and southwest of GW-87. If the simulations significantly overestimate concentrations in this area, recovery could be more effective in containing the plumes than the modeling predicts. However, if there are significant quantities of NAPLs in the saturated zone the main effects on the recovery well scenarios would be an increase in concentrations of VOCs in the extracted water and a reduction in the overall efficiency of recovery as discussed below.

### 7.3.4.4 Capping with source terms

As alluded to earlier the results of capping simulations with source terms, which represent long-term NAPL sources, are virtually indistinguishable from the results of the no-action simulations. The only difference between model input to capping-with-source simulations and no-action simulations is the loss of recharge over the area of the cap. This loss amounts to only about 6% of total recharge over the modeled area and does not have a significant effect on transport simulation results. Consequently, Figs. 7.4–7.12 can be considered to represent the results of the capping with source-terms simulations, and the discussion of the results in Sect. 7.4.3.1 applies as well. Because some diminution of contaminant input would be expected over time, these results probably resemble worst-case results.

Fig. 7.19. PCE distribution at 15 years from 1988 for the 600-ft recovery alternative without source terms. Concentrations in $\mu g/L$. Highest contour 200 $\mu g/L$.

Fig. 7.20. PCE distribution at 30 years from 1988 for the 600-ft recovery alternative without source terms. Concentrations in $\mu g/L$. Highest contour 100 $\mu g/L$.

Fig. 7.21. TCE distribution at 15 years from 1988 for the 600-ft recovery alternative without source terms. Concentrations in $\mu g/L$. Highest contour 200 $\mu g/L$.

**Fig. 7.22.** TCE distribution at 30 years from 1988 for the 600-ft recovery alternative without source terms. Concentrations in μg/L. Highest contour 50 μg/L.

Fig. 7.23. 1-1-DCE distribution at 15 years from 1988 for the 600-ft recovery alternative without source terms. Concentrations in $\mu g/L$.

Fig. 7.24. 1-1-DCE distribution at 30 years from 1988 for the 600-ft recovery alternative without source terms. Concentrations in $\mu g/L$.

64



Fig. 7.25. PCE distribution at 15 years from 1988 for the 200-ft recovery alternative without source terms. Concentrations in μg/L. Highest contour 100 μg/L.

65
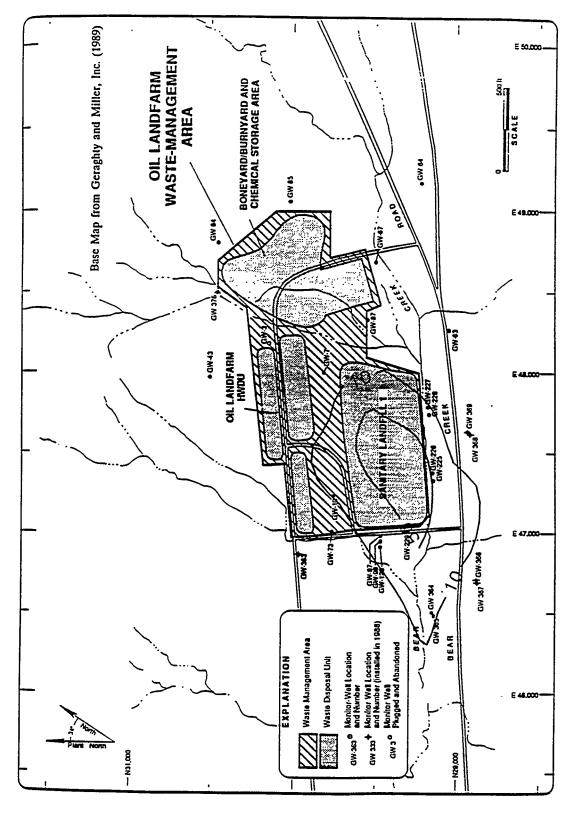


Fig. 7.26. PCE distribution at 30 years from 1988 for the 200-ft recovery alternative without source terms. Concentrations in μg/L.
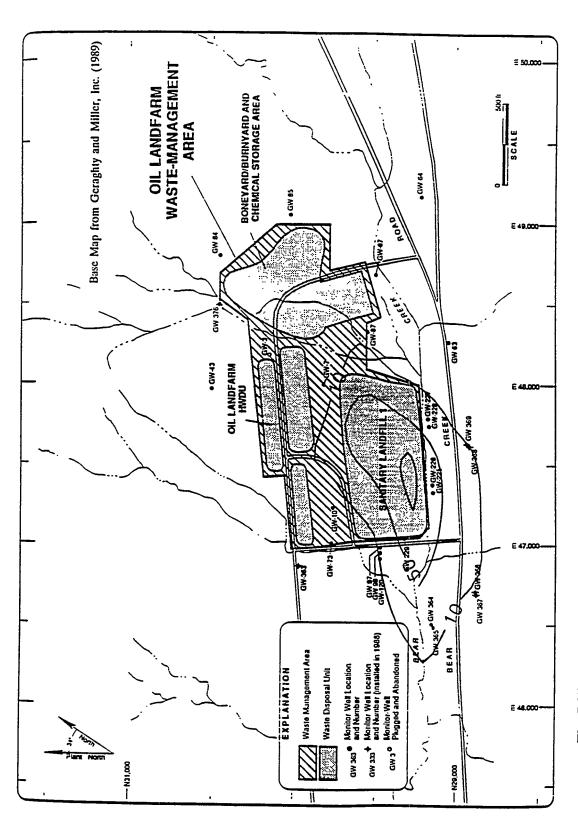
Fig. 7.27. TCE distribution at 15 years from 1988 for the 200-ft recovery alternative without source terms. Concentrations in $\mu g/L$. Highest contour 100 $\mu g/L$.
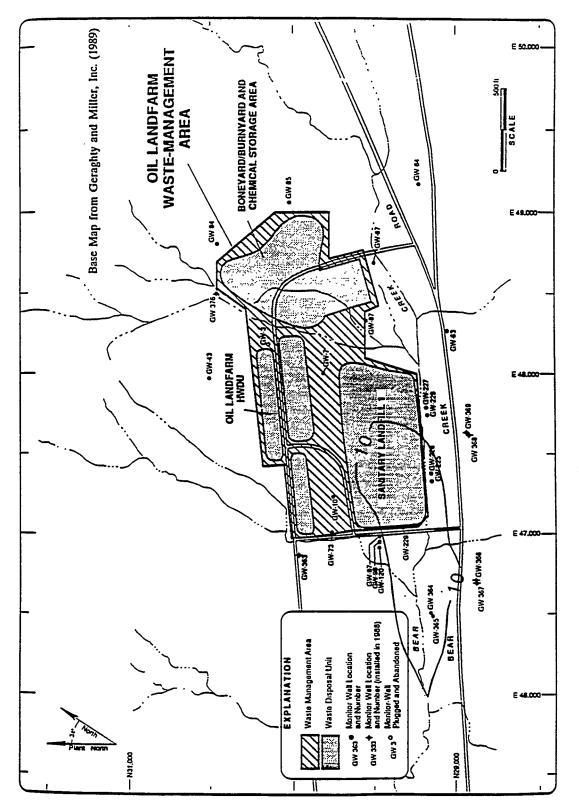
Fig. 7.28. TCE distribution at 30 years from 1988 for the 200-ft recovery alternative without source terms. Concentrations in μg/L.
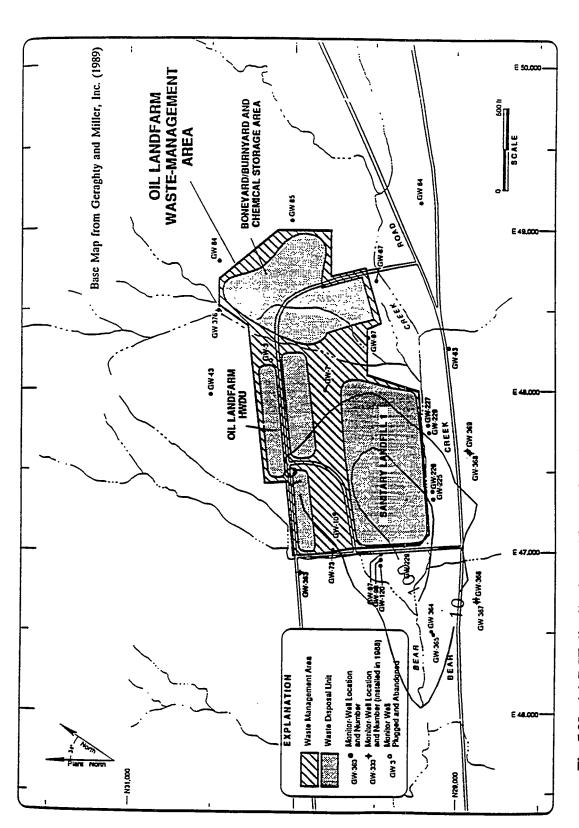
Fig. 7.29. 1,1-DCE distribution at 15 years from 1988 for the 200-ft recovery alternative without source terms. Concentrations in $\mu g/L$.

### 7.3.4.5 Capping and recovery wells with source terms

Figures 7.30–7.32 show the results of recovery well simulations with source terms for the 600-ft case at 30 years from 1988, and Figs. 7.33–7.35 show the results for the 200-ft case at the same time. Comparison of these figures with the 30-year figures for capping with source terms (i.e., no action) shows that the simulations predict that recovery in the 600-ft case is only slightly more effective than capping in reducing contaminant concentrations and plume areas. The 200-ft case is more efficient and reduces plume areas and peak concentrations by about a factor of 2 for all contaminants. Although in both cases the plumes persist indefinitely because of continuous contaminant input from the source terms, plume migration is limited to less than 2000 ft from the southwest edge of the WMA. In both cases the plumes reach steady-state by about 30 years because contaminant input from the source terms is balanced by contaminant discharge to the recovery wells and to the Bear Creek sinks; in the 200-ft case the wells remove a greater proportion of the mass than in the 600-ft case.

## 7.4 CONTAMINANT MIGRATION AND PERSISTENCE IN SURFACE WATER, SOILS, AND SEDIMENTS

Movement of the VOCs via soils and sediments is not an important migration pathway when compared to transport in groundwater and surface water. As discussed earlier, these compounds are heavier than water and tend to sink into the subsurface where they are isolated from contact with soils. In the subsurface they act as sources of contamination to groundwater and, via groundwater discharge, to surface water. Once in surface water these contaminants rapidly volatilize.

Bear Creek in the vicinity of the site is not currently contaminated with VOCs, but, as the contaminant plumes migrate farther from their sources and discharge larger amounts of contaminants to the creek concentrations in the creek could rise to levels above standards. Because of the lack of data needed to calibrate calculations, we did not attempt to predict time- and distance-dependent surface water concentrations for this site. However, from previous experience with surface water calculations for the Bear Creek Burial Grounds we would expect contaminant concentrations in surface water to be roughly an order of magnitude below those found in the groundwater discharging into the surface water. Thus even peak no-action or capping with source-terms VOC concentrations in the creek south of the WMA should be only in the low 10s of $\mu g/L$, and should fall off very rapidly with distance downstream from the site due to dilution and volatilization. Peak concentrations for other alternatives can be expected to be even less.

Much of the transport of PCBs (and a minor component of the transport of VOCs) to streams occurs as a result of the movement of oil-contaminated soils and debris, as well as floating oil films, following moderate to heavy rainfall. Because of their solubility and sorption properties, PCBs tend to remain in the sediment in surface water systems. Large volumes of PCB-contaminated soils have already been removed from the Oil Landfarm disposal plots, and capping of the site is well under way. These measures should eliminate erosion of the soils and subsequent movement with sediments as a sources of contaminant transport. Implementation of the groundwater withdrawal system and subsequent elimination of permanent surface flow in Tributary 3 would further reduce the likelihood of contaminant transport.
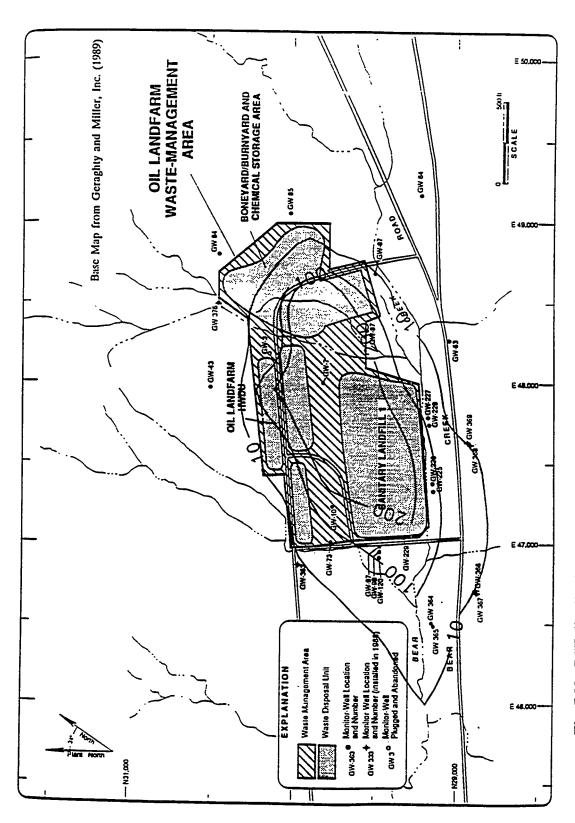
Fig. 7.30. PCE distribution at 30 years from 1988 for the 600-ft recovery alternative with source terms. Concentrations in μg/L.
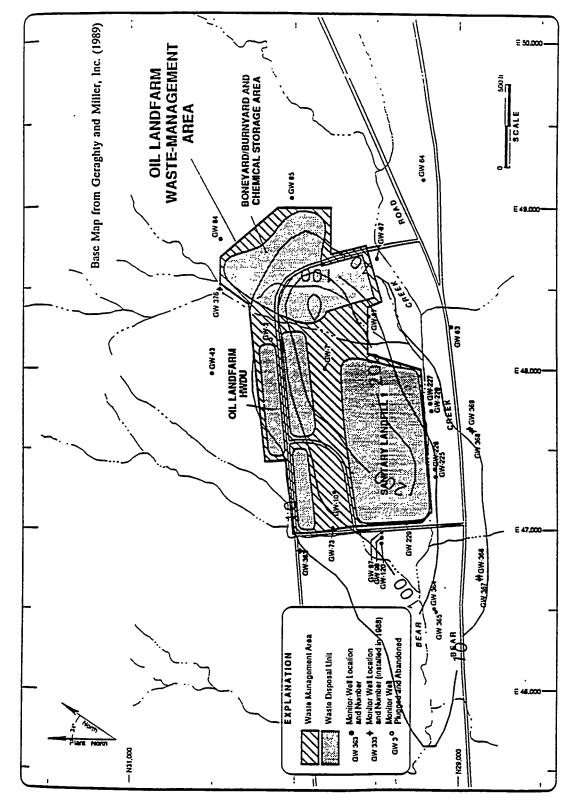
Fig. 7.31. TCE distribution at 30 years from 1988 for the 600-ft recovery alternative with source terms. Concentrations in $\mu g/L$.
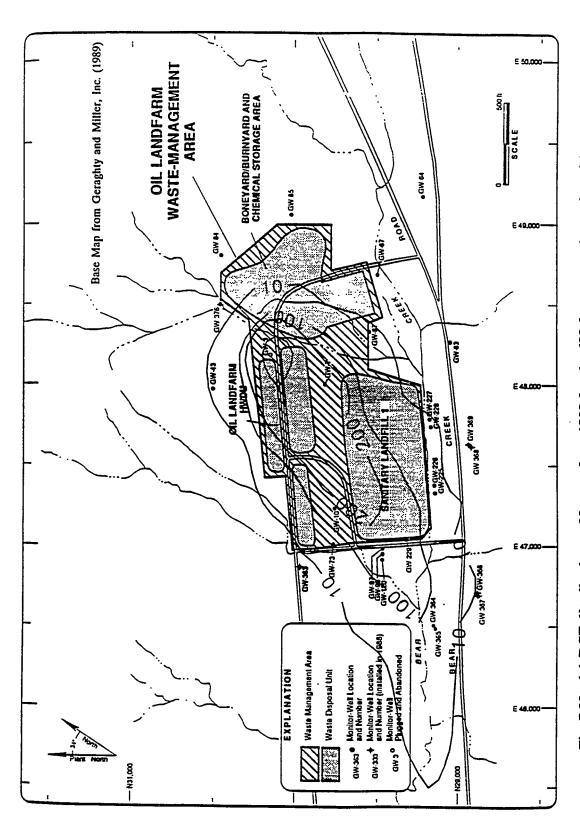
**Fig. 7.32.** 1,1-DCE distribution at 30 years from 1988 for the 600-ft recovery alternative with source terms. Concentrations in $\mu g/L$. Highest contour 500 $\mu g/L$.
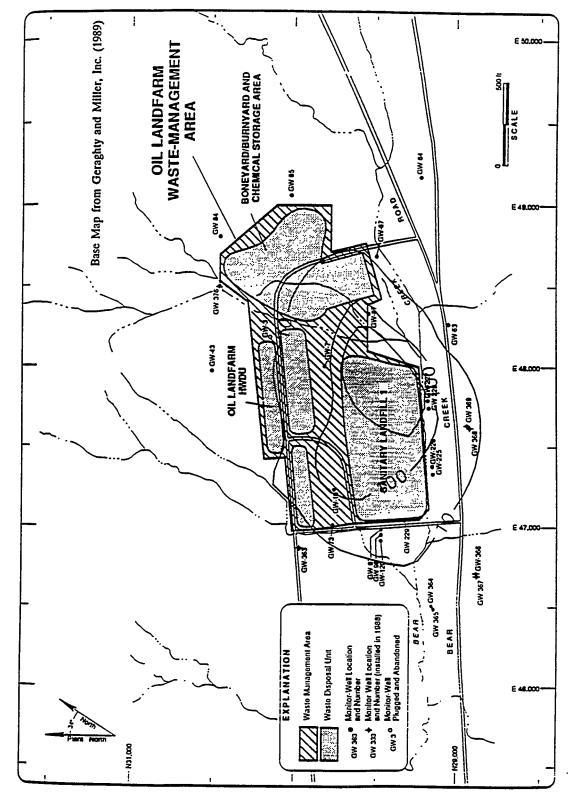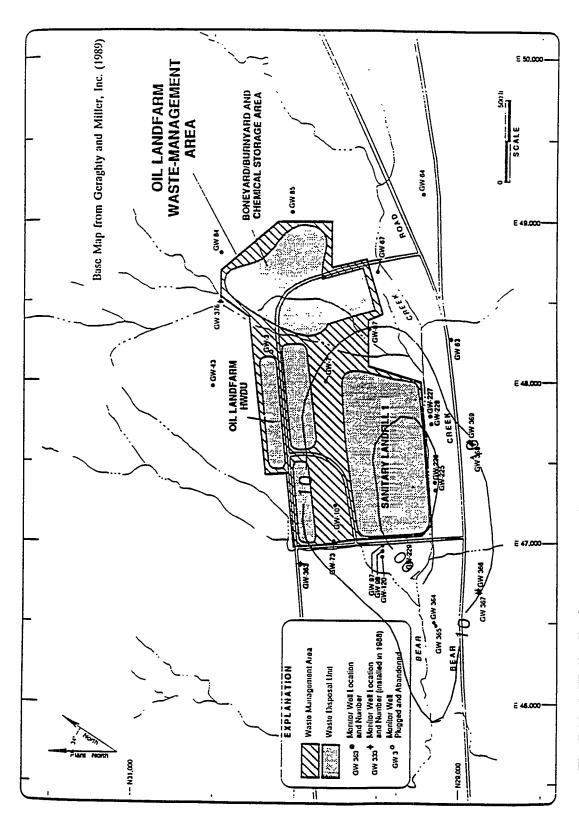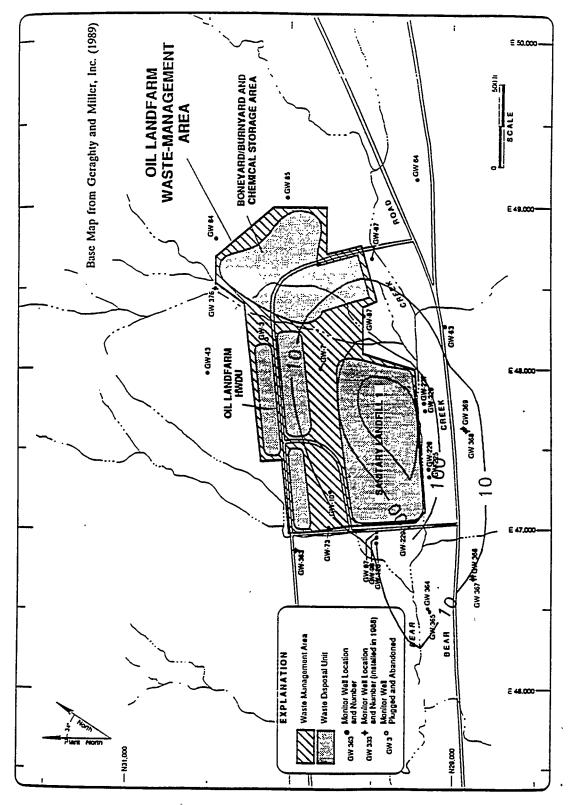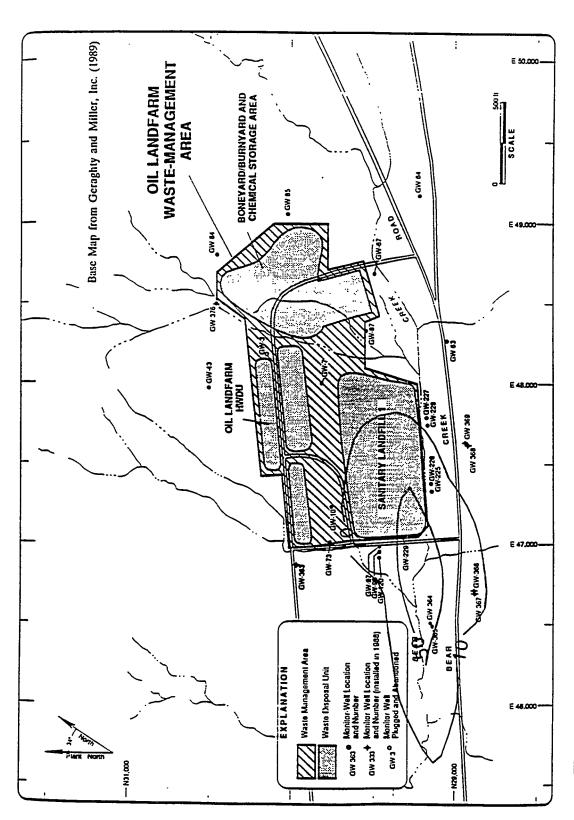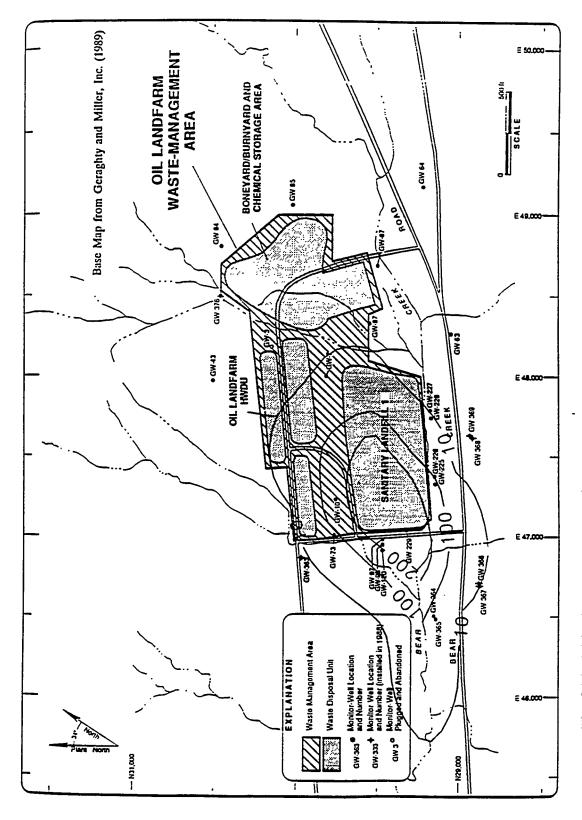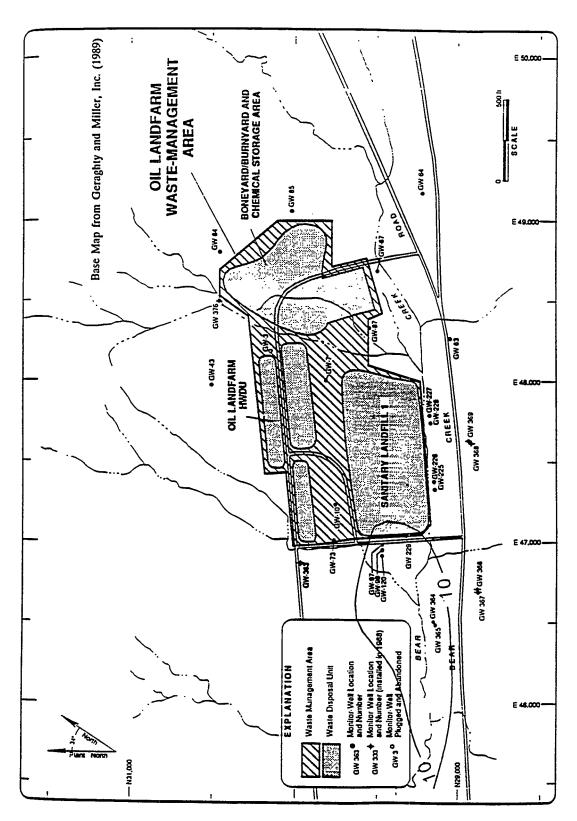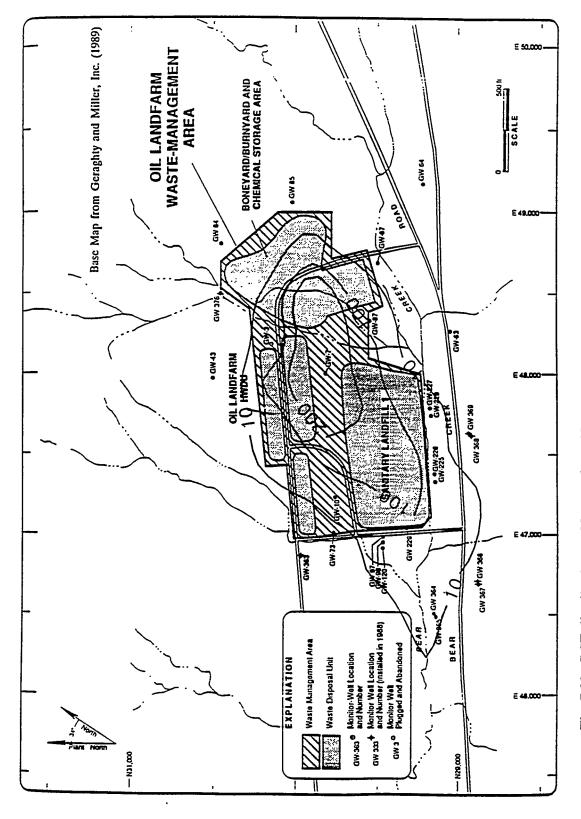
Fig. 7.33. PCE distribution at 30 years from 1988 for the 200-ft recovery alternative with source terms. Concentrations in µg/L.
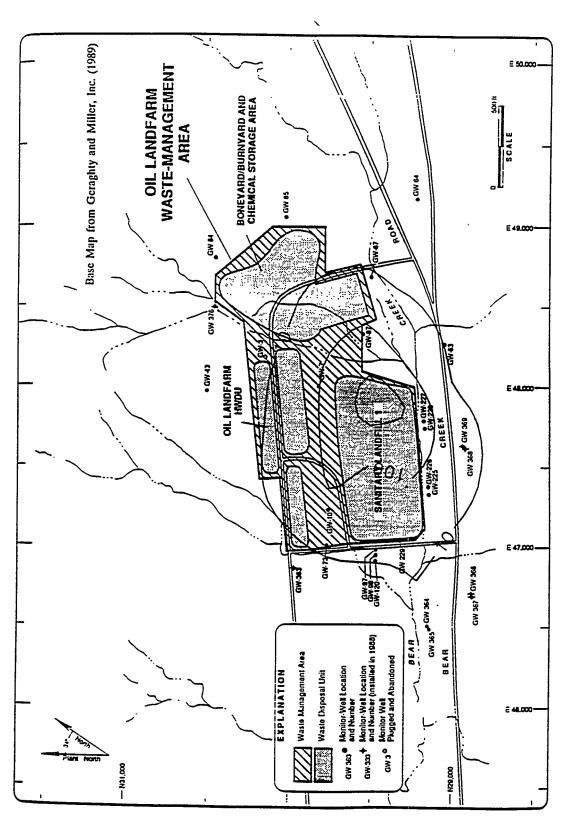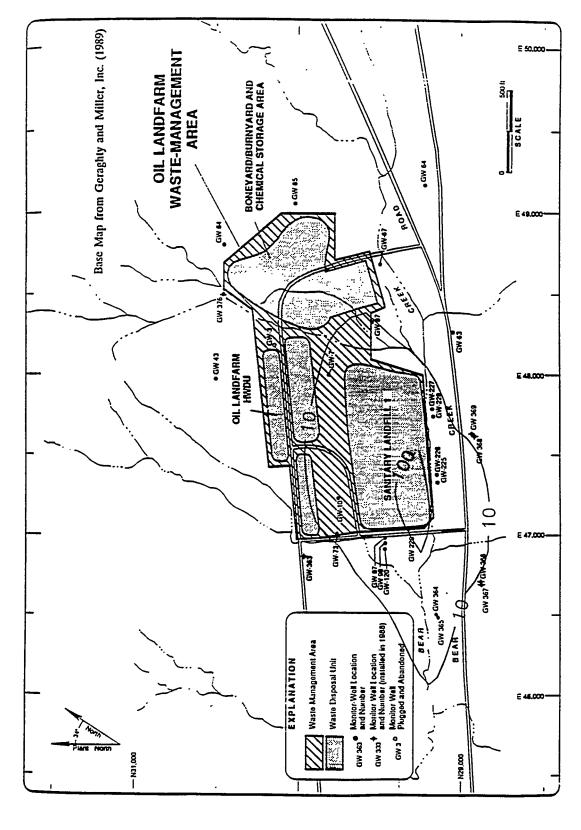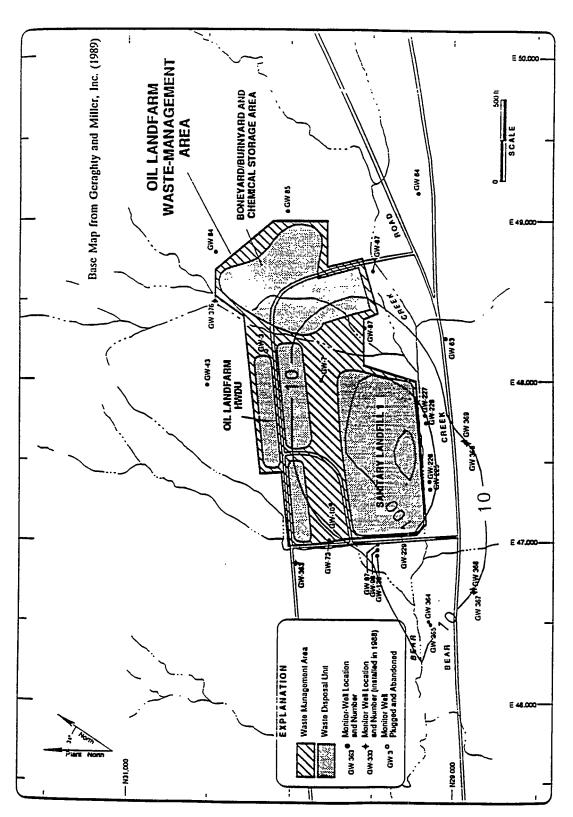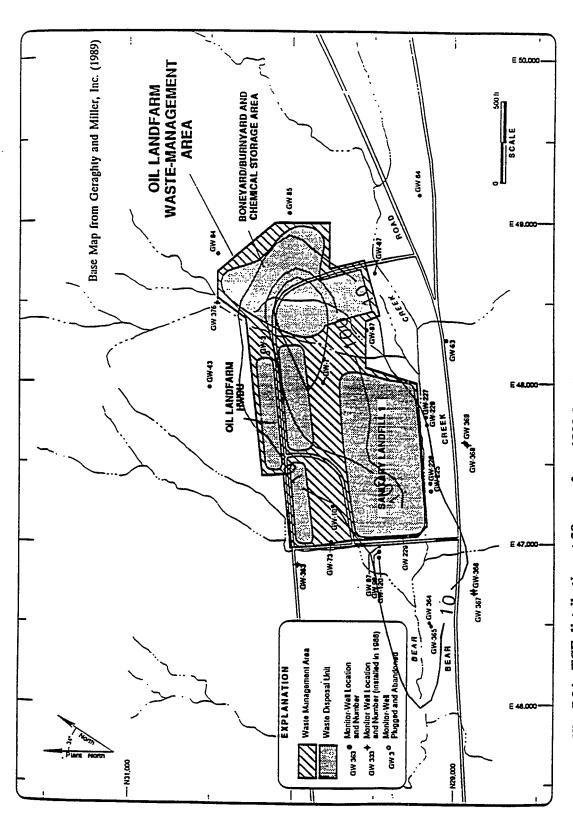
Fig. 7.34. TCE distribution at 30 years from 1988 for the 200-ft recovery alternative with source terms. Concentrations in $\mu g/L$. Highest contour 200 $\mu g/L$.
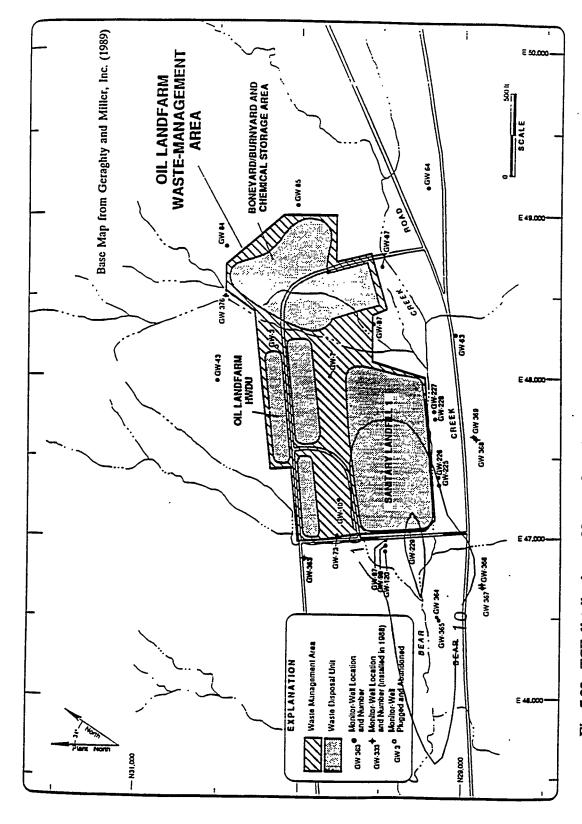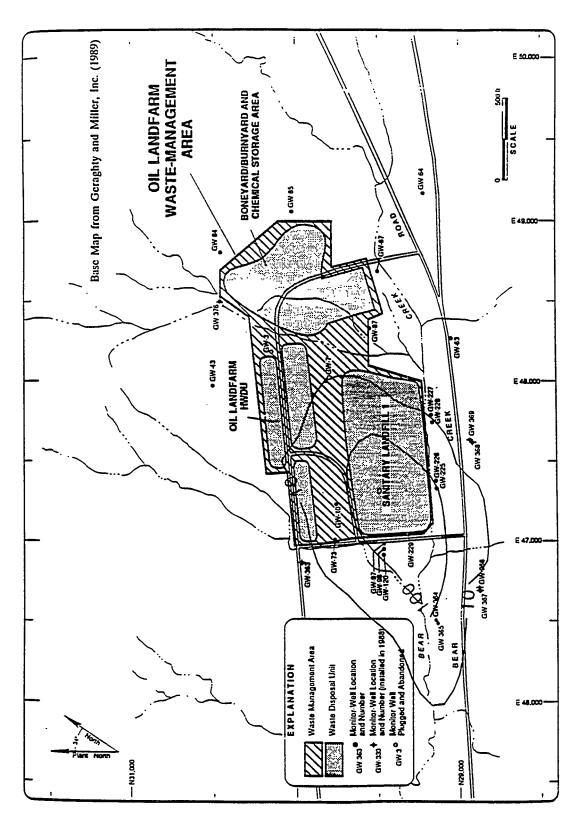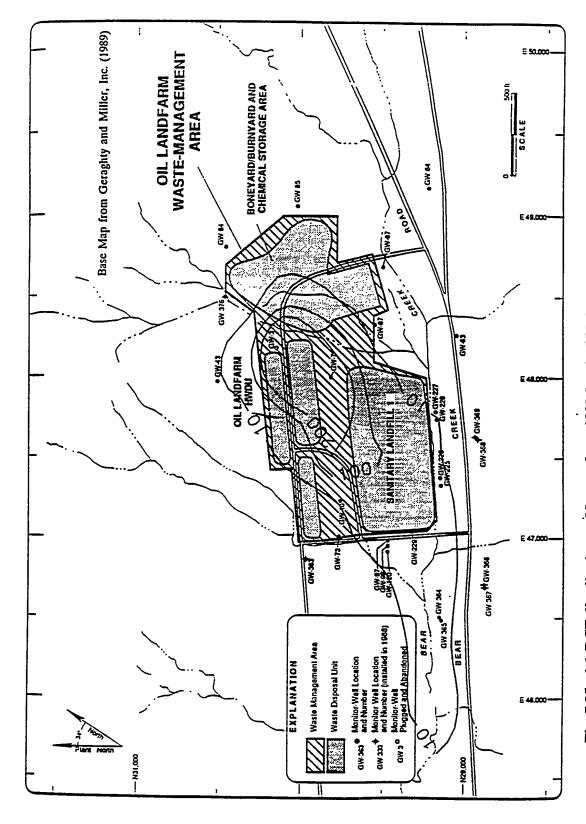
Fig. 7.35. 1,1-DCE distribution at 30 years from 1988 for the 200-ft recovery alternative with source terms. Concentrations in $\mu g/L$.
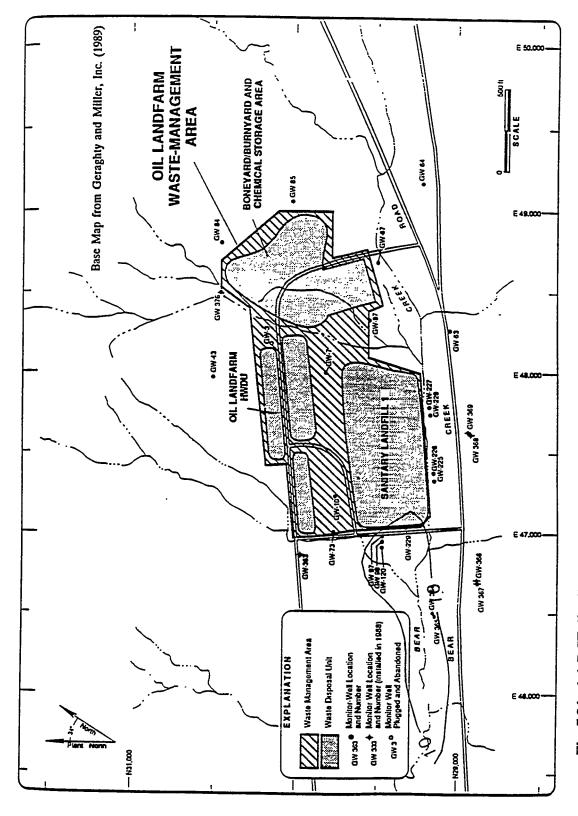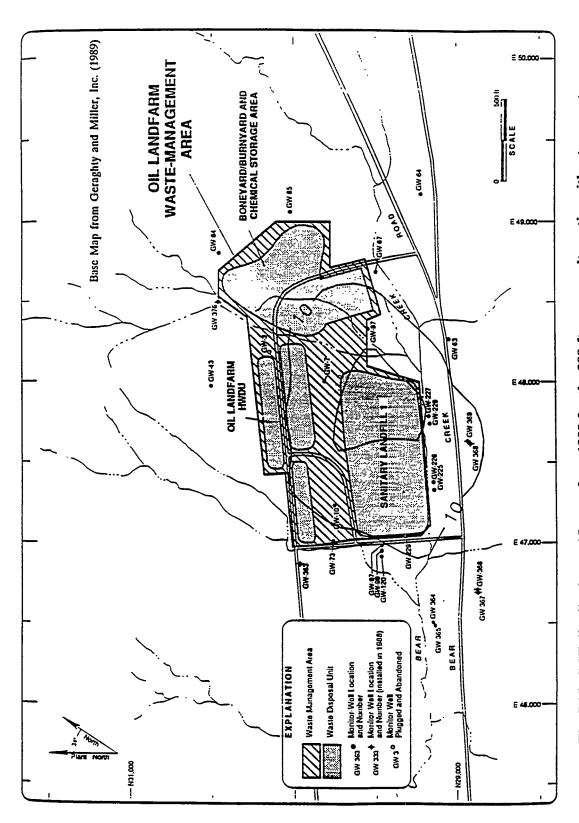
## 7.5 AIRBORNE CONTAMINANT TRANSPORT AND EXPOSURE

### 7.5.1 Introduction

Groundwater extracted from the Oil Landfarm WMA site is to be transported to the Bear Creek Burial Grounds site, where it will be treated together with groundwater from that site in the Groundwater Treatment Facility (GWTF). Operation of the GWTF at the Bear Creek Burial Grounds site would result in discharge to the atmosphere of contaminants removed form the treated water. These contaminants would then be dispersed by prevailing winds over the Y-12 Plant and environs, possibly exposing plant workers and members of the general public to airborne concentrations of the contaminants.

Estimates of potential airborne concentrations of and resulting exposures to four contaminants resulting from operation of the GWTF are presented in Walter et al. (1990). The four contaminants considered in that study are tetrachloroethene, trichloroethene, trans-1,2-dichloroethene, and 1,1-DCE. Those estimates are based on groundwater input to the GWTF solely from the Burial Grounds site, and are calculated for locations within the Y-12 Plant boundary, at selected off-site locations, and within an 80-km (50-mile) radius of the plant center. In this report, we present airborne concentration and exposure estimates for the same four contaminants at the same locations resulting from the combined input of groundwater from the Burial Grounds and the Oil Landfarm WMA to the GWTF. Although trans-1,2-dichloroethene is not an indicator chemical for the Oil Landfarm site, recovery well simulations were performed for this compound to estimate its contribution to GWTF off-gas.

Population, meteorological, and contaminant data are input to the Inhalation Exposure Methodology (IEM) (O'Donnell 1983; O'Donnell and Cooper 1986) to estimate airborne contaminant concentrations and estimates of individual and collective exposures to those contaminants. An airborne concentration is defined as the annual average mass of a contaminant contained in a unit volume of air at or near ground level over a specified location. Concentrations are expressed in units of micrograms per cubic meter ($\mu$g/m$^3$) or picograms per cubic meter (pg/m$^3$), where 1 $\mu$g = 10$^{-6}$ g, and 1 pg = 10$^{-6}$ $\mu$g = 10$^{-12}$ g.

An exposure estimate is a measure of human contact with a contaminant. Exposures are the product of an airborne concentration and the number of people immersed in that concentration; thus, they are expressed in units of person $\mu$g/m$^3$ or person pg/m$^3$. Two types of exposure are considered, individual and collective. An individual exposure is numerically equal to the airborne concentration to which the designated individual is exposed. A collective exposure is the sum of exposures to all individuals residing in the region of interest. In practice, a collective exposure is obtained by dividing the region of interest into smaller areas, apportioning the population residing in the region among the smaller areas, determining an average airborne concentration over each area, multiplying the average concentration by the number of people assigned to each area, and adding the results for each area.

### 7.5.2 Description of Atmospheric Transport and Exposure Estimation Models

Airborne concentrations of and exposures to the four contaminants were estimated using the IEM. This methodology uses the long-term version of the Industrial Source Complex Dispersion Model (a steady-state, Gaussian-plume, atmospheric dispersion model) to calculate

annual average, ground-level air concentrations of atmospherically released contaminants at selected locations (H. E. Cramer Company, Inc. 1985; Bowers et al. 1979). The CONEX code of the IEM combines these concentrations with population distribution data to give estimates of individual and collective exposures.

Data input to the atmospheric dispersion model include meteorological data collected at the Y-12 Plant during calendar year 1986 (Oakes et al. 1987), engineering estimates of source parameter values, and the locations for which concentrations and exposures are to be calculated. The treatment plant is located ~4960 m southwest (3810 m west by 2743 m south) of an arbitrarily chosen Y-12 Plant center (see Fig. 7.39). The plant is modeled as a stack (point) source with a release height of 30 ft (9.14 m), an inner diameter of 10 in (0.25 m), an exit gas velocity of 30 ft/s (9.14 m/s), and an ambient exit gas temperature of 294 degrees Kelvin (K). Contaminant release rates are estimated from the treatment module characteristics and input water concentrations predicted by the groundwater modeling. The predicted mass flux of contaminants in groundwater extracted from the Oil Landfarm WMA is added to that from the Burial Grounds. The airborne contaminant emission rates (Table 7.2) for combined treatment, which reflect this addition, show that contributions of contaminants from the Oil Landfarm WMA are 22% for 1,1-DCE, 6% for TCE, 1% for PCE, and negligible for trans-1,2-dichloroethene.

Table 7.2. Airborne contaminant emission rates for the proposed
Bear Creek Burial Grounds Groundwater Treatment Facility

| Contaminant | Emission rate (g/s) | |
| --- | --- | --- |
| | Burial grounds | Burial grounds and oil landfarm |
| Tetrachloroethene | 0.0210 | 0.0212 |
| Trichloroethene | 0.0029 · | 0.0031 |
| trans-1,2-Dichloroethene | 0.0067 | 0.0067 |
| 1,1-Dichloroethene | 0.00073 | 0.00093 |

Two coordinate systems, with their origins at the arbitrary plant center, are used in the concentration and exposure calculations. A rectangular $(X,Y)$ system is used to calculate concentrations and individual exposures within and immediately adjacent to the Y-12 Plant (Fig. 7.36). A polar grid (not shown) is used to calculate average concentrations and collective exposures between the Y-12 Plant boundary and an 80-km (50-mile) radius.

The rectangular $(X,Y)$ grid is positioned so that its $X$ direction runs from west (~6000 m) to east (+2000 m) and its $Y$ direction runs from south (~3000 m) to north (+1600 m) in 200-m increments. Airborne contaminant concentrations are calculated at each of the 984 grid nodes. These concentrations are used to estimate individual exposures within and near the Y-12 Plant boundary and to construct the concentration contours shown in Figs. 7.37–7.40.

The polar grid consists of 160 segments (small areas). The center of a segment is located at the intersection of one of 16 radial lines and one of 10 concentric rings. Each radial line lies along a major compass direction (e.g., north). The rings are located 1,207, 2,414, 4,023,
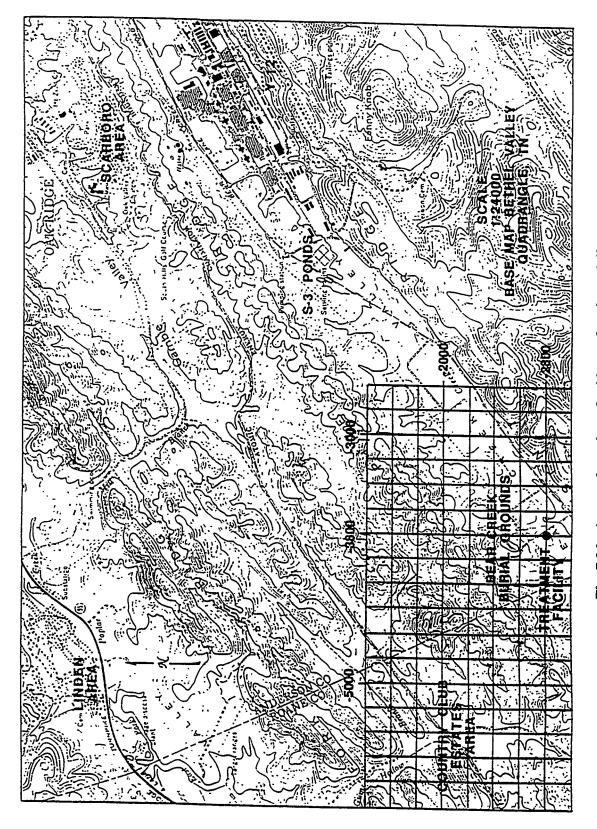
Fig. 7.36. Area and portion of grid map for air modeling.
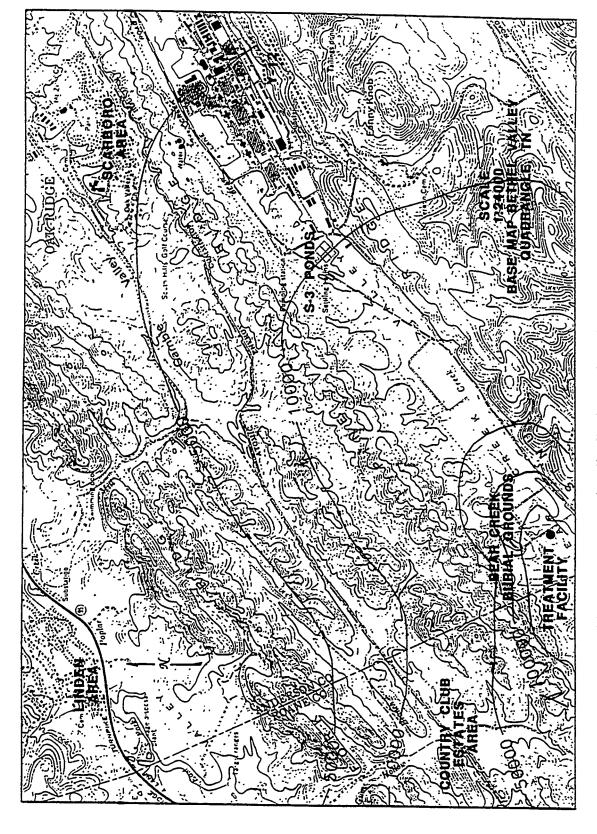
**Fig. 7.37. PCE ground-level air concentration distribution at time of groundwater treatment facility startup (1997). Concentrations in pg/m³.**
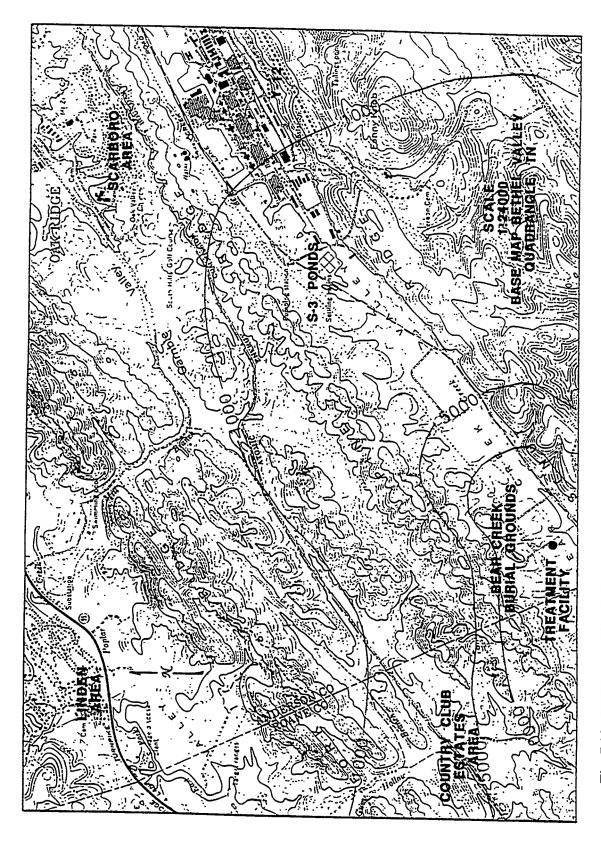
Fig. 7.38. TCE ground-level air concentration distribution at time of groundwater treatment facility startup (1997). Concentrations in pg/m³.
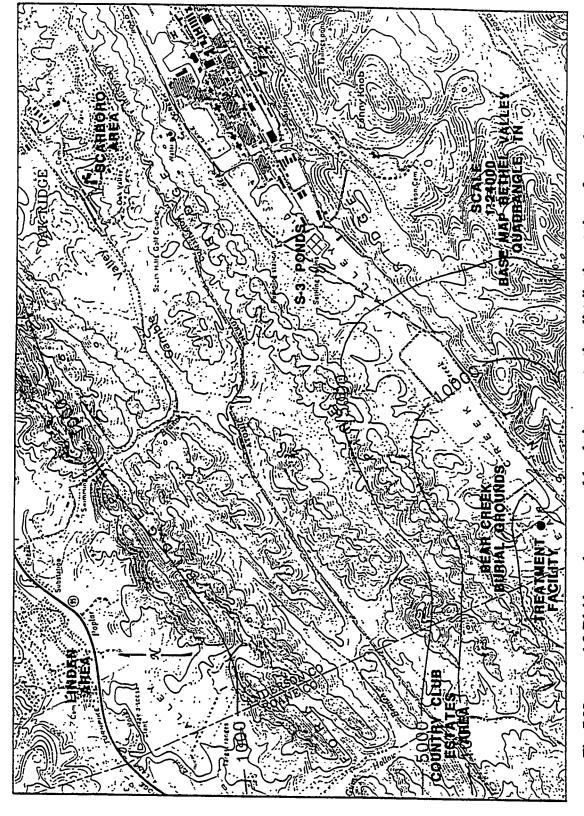
**Fig. 7.39.** *trans*-1,2-Dichloroethene ground-level air concentration distribution at time of groundwater treatment facility startup (1997). Concentrations in $pg/m^3$.
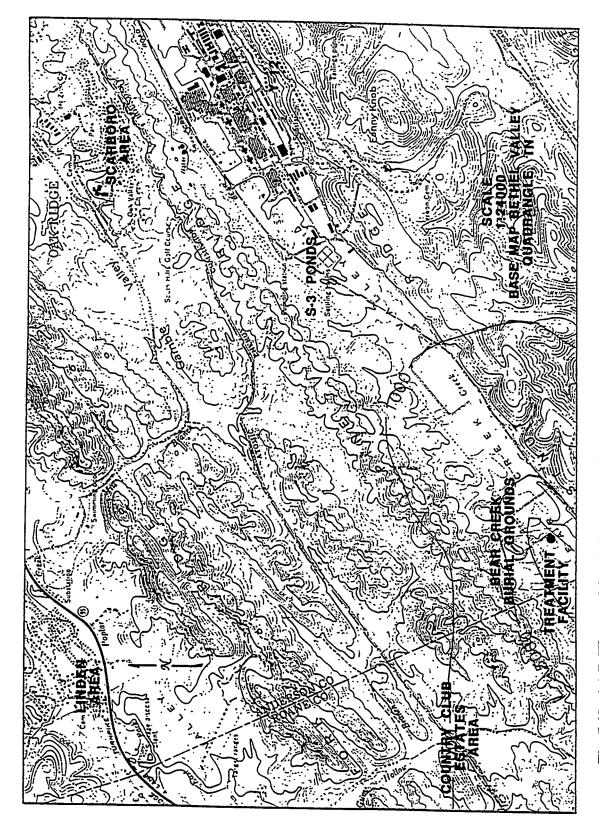
Fig. 7.40. 1,1-DCE ground-level air concentration distribution at time of groundwater treatment facility startup (1997). Concentrations in pg/m$^3$.

5,633, 7,242, 12,070, 24,140, 40,234, 56,327, and 72,420 m from the plant center. The area covered by a segment is bounded by radial lines lying ±11.25 degrees from its compass direction and by rings located midway between the rings that define its center and the centers of the two adjoining segments. Average airborne contaminant concentrations are calculated at each of the 160 segment centers. The ~863,000 people residing within 80 km (50 miles) of the plant (based on 1980 Census data) are apportioned among the segments of the grid (Table 7.3). The collective exposure within a segment is estimated by multiplying its average contaminant concentration by the number of people residing in the segment. Total collective exposures are obtained by summing all segment exposures.

## 7.5.3 Modeling Results

As expected, the highest calculated airborne contaminant concentrations and, hence, potential individual worker exposures occur adjacent to the treatment module (see Table 7.4 and Figs. 7.37–7.40). Concentrations and potential individual worker exposures decrease rapidly with distance from the GWTF. Potential worker exposures over the main plant area are relatively small because the wind blows mostly to the west and southwest, away from the main plant area, and because the of the distance between the GWTF and the plant. Because airborne concentrations are directly proportional to release rates, the contribution of the Oil Landfarm WMA to contaminant levels and potential worker exposures is relatively small.

The maximum off-site individual exposure is predicted at a residence in Country Club Estates ($X$-$Y$ coordinates = -6000, -2300) which lies in the predominant wind direction from the treatment plant. In fact, the Country Club Estates area is subject to the highest off-site concentrations and potential individual public exposures (Table 7.5 and Figs. 7.37–7.40). Other populated areas should be subjected to much lower concentrations.

Table 7.6 is a summary of off-site potential collective exposures and maximum and potential average individual exposures within 50 km of the Y-12 Plant. Figure 7.41 is based on the results of the polar grid-based modeling and illustrates the average ground-level air concentrations as a function of distance downwind (WSW) from the center of the Y-12 Plant. Concentrations approximately 1 km downwind from the facility are about an order of magnitude less than those immediately above the facility. Again contaminants from the Oil Landfarm make relatively small contributions to airborne concentration levels.

## Table 7.3. Number of people residing in each segment

| Direction | Distance to ring (m) | | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|
| | 1,207 | 2,414 | 4,023 | 5,633 | 7,242 | 12,070 | 24,140 | 40,234 | 56,327 | 72,420 |
| N | 1,352 | 1,362 | 1,613 | 0 | 1,027 | 0 | 5,246 | 6,879 | 5,599 | 9,855 |
| NNE | 0 | 0 | 1,303 | 1,481 | 1,616 | 2,885 | 8,784 | 14,072 | 7,409 | 7,317 |
| NE | 1,765 | 0 | 0 | 0 | 0 | 0 | 8,063 | 5,687 | 5,521 | 12,371 |
| ENE | 0 | 0 | 1,466 | 0 | 0 | 3,442 | 23,779 | 17,192 | 11,126 | 16,563 |
| E | 0 | 0 | 0 | 0 | 0 | 5,647 | 142,878 | 51,834 | 13,173 | 11,229 |
| ESE | 0 | 0 | 0 | 0 | 0 | 11,420 | 42,654 | 19,424 | 12,319 | 14,390 |
| SE | 0 | 0 | 0 | 0 | 842 | 1,976 | 13,084 | 44,646 | 2,269 | 1,331 |
| SSE | 0 | 0 | 0 | 0 | 0 | 4,128 | 6,635 | 9,040 | 1,728 | 2,231 |
| S | 0 | 0 | 0 | 0 | 0 | 0 | 8,494 | 5,867 | 6,549 | 4,275 |
| SSW | 0 | 0 | 0 | 0 | 0 | 10,014 | 7,741 | 9,381 | 14,508 | 25,477 |
| SW | 0 | 0 | 0 | 0 | 0 | 11,640 | 3,785 | 4,155 | 5,692 | 8,546 |
| WSW | 0 | 0 | 0 | 0 | 0 | 0 | 17,811 | 12,096 | 5,965 | 5,397 |
| W | 0 | 1,505 | 0 | 0 | 0 | 1,229 | 7,584 | 4,008 | 5,309 | 14,117 |
| WNW | 0 | 0 | 3,149 | 2,105 | 0 | 3,101 | 3,153 | 2,536 | 2,845 | 4,337 |
| NW | 0 | 0 | 3,094 | 623 | 0 | 1,500 | 1,863 | 2,310 | 3,689 | 5,488 |
| NN | 0 | 0 | 3,585 | 0 | 0 | 1,666 | 1,071 | 2,866 | 8,940 | 6,531 |

Table 7.4. Summary of calculated on-site air concentrations (pg/m³)
and potential individual worker exposures (person-pg/m³)

| Location | Tetra-chloroethene | Tri-chloroethene | *trans*-1,2-Di-chloroethene | 1,1-Di-chloroethene |
|---|---|---|---|---|
| Near treatment facility | 486,100 | 70,522 | 153,710 | 21,339 |
| S-3 Ponds | 10,978 | 1,593 | 3,470 | 484 |
| Main plant: | | | | |
| West end | 12,945 | 1,876 | 4,090 | 573 |
| Center | 4,995 | 721 | 1,580 | 217 |
| East end | 3,148 | 456 | 990 | 140 |

Table 7.5. Summary of calculated off-site air concentrations (pg/m³)
and potential individual exposures (person-pg/m³)

| Location | Tetra-chloroethene | Tri-chloroethene | *trans*-1,2-Di-chloroethene | 1,1-Di-chloroethene |
|---|---|---|---|---|
| Maximum residence Country Club | 32,429 | 4,709 | 10,250 | 1,422 |
| Estates | 6,942–36,798 | 1,007–5,339 | 2,200–11,640 | 305–1,613 |
| Linden School | 1,635 | 233 | 520 | 72 |
| Linden area | 1,261–2,583 | 180–371 | 400–820 | 51–114 |
| Scarboro area | 3,289–5,005 | 477–731 | 1,040–1,580 | 140–216 |

Table 7.6. Summary of potential exposures to contaminants that might be released from
the proposed groundwater treatment plant at the Bear Creek Burial Grounds
(includes input from both the Burial Ground and the Oil Landfarm)

| Contaminant | Individuals (g/m³) | | Collective (person-µg/m³) |
|---|---|---|---|
| | Maximum | Average | |
| Tetrachloroethene | 32,492 | 472 | 408 |
| Trichloroethene | 4,706 | 69 | 59 |
| *trans*-1,2-Dichloroethene | 10,250 | 149 | 129 |
| 1,1-Dichloroethene | 1,422 | 20 | 18 |
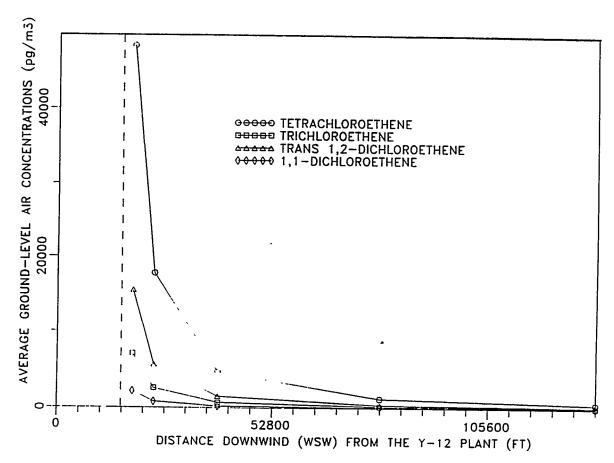
Fig. 7.41. Average ground-level air concentration as a function of distance downwind from the center of the Y-12 Plant (WSW) at time of groundwater treatment facility startup (1997). Dashed line marks the location of the groundwater treatment facility.

# 8. RISK ASSESSMENT: HUMAN HEALTH

## 8.1 INTRODUCTION

The following sections present a health risk assessment for each remedial alternative for the migration pathways considered in Sect. 7. For groundwater and air the assessments are based on the results of computer modeling as described in Sect. 7 and in Appendix A. The health risks associated with groundwater are treated quantitatively by calculating hazard indexes and excess cancer risks for each medium at selected locations and times. The definitions of these risk indicators and the rationale for selecting locations are described below. Health risks related to air emitted from the treatment modules are evaluated by comparing predicted concentrations at selected locations with acceptable concentrations, based on the American Conference of Governmental Industrial Hygienists' Threshold Limit Values (TLVs) or on excess cancer risks. Risks posed by surface water are evaluated qualitatively. The risks presented by soils and sediments are negligible compared to the other media given the remedial measures already under way (see Sect. 7.4), and are not discussed further.

## 8.2 TOXICOLOGICAL PROPERTIES OF CONTAMINANTS

### 8.2.1 Tetrachloroethene

Tetrachloroethene (PCE) is toxic to the liver and kidneys and is a central nervous system depressant. Metabolism of PCE may enhance its toxicity; however, humans have limited abilities to metabolize PCE, and most systemically absorbed PCE is eliminated unmetabolized. PCE is readily absorbed through the lungs, skin, and gastrointestinal tract. It is one of the least hepatotoxic and nephrotoxic of the chlorinated hydrocarbons. Residual organ damage is not commonly observed in humans who have been exposed to large quantities of the compound. Results of carcinogenicity studies with rats and mice are contradictory; however, PCE has been designated by EPA as a class B2 carcinogen (Arthur D. Little, Inc., 1985).

### 8.2.2 Trichloroethene

Trichloroethene (TCE) is a central nervous system depressant. Clinical symptoms of exposure include gastric distress, narcosis, and cardiac abnormalities. Trichloroethene is readily metabolized in the human body but has a limited capacity for residual tissue damage. Acute exposure to TCE depresses the central nervous system, causing such symptoms as headache, dizziness, vertigo, tremors, irregular heartbeat, fatigue, nausea, vomiting, blurred vision, and intoxication similar to that of ethanol. The vapors may cause irritation of the eyes, nose, and throat. The liquid may cause burning irritation and damage to the eyes. Repeated or prolonged skin contact with the liquid may cause dermatitis.

Results of studies designed to evaluate the carcinogenicity and mutagenicity of trichloroethene are inconclusive. Opinion is divided as to whether the carcinogenic and mutagenic effects observed result from trichloroethene conversion or are attributable to contaminants in trichloroethene formulations. TCE has been designated by EPA as a class B2 carcinogen (Arthur D. Little, Inc. 1985).

### 8.2.3 1,1-Dichloroethene

1,1-Dichloroethene is a central nervous system depressant and is toxic to the liver and kidneys. Inhalation studies using small mammals exposed for a 6-month period showed significant mortality and liver damage when exposed to levels as high as 47 ppm. There was minimal injury to the liver at concentrations of 25 ppm, and no adverse effects were exhibited at dose levels of 6 to 8 ppm. In liquid form, the chemical is moderately irritating to the eyes, causing pain, conjunctival irritation, and possible transient injury. The liquid is irritating to the skin after only a few minutes contact. Results of carcinogenicity studies in rats and mice are inconclusive; however, 1,1-dichloroethene is designated by EPA as a class C carcinogen (Arthur D. Little, Inc., 1985).

### 8.2.4 Polychlorinated Biphenyls

Polychlorinated biphenyls (PCBs) are a family of man-made chemicals containing 209 compounds. In humans, PCBs are found in fat, blood, and breast milk. PCBs primarily affect the liver and cutaneous tissue. Inhalation and dermal absorption through contact with contaminated soil are likely to be major routes of exposure. Eye, nose, and throat irritations are common in occupational settings. Gross toxic effects observed in occupational settings are centered around skin lesions, especially chloracne, sebaceous cysts, and pustules. Liver damage, digestive irritations, and eye irritation are also possible. The major biochemical effect is on the liver, which exhibits elevated levels of enzyme production. Both acute and chronic exposure have resulted in edema, jaundice, vomiting, anorexia, nausea, abdominal pains, and fatigue. Severe liver damage has resulted in coma and, in some cases, death. Animal studies indicate that PCBs are fetotoxic in rats, mice, rabbits, and monkeys. Reproductive effects have been observed in monkeys, mink, and, at high doses, in rodents. In addition to effects on the liver and cutaneous tissues, animal studies have indicated that stomach and thyroid alterations, immunosuppression, and porphyria may occur in certain animals. The carcinogenic end point for PCBs in animals is the liver, but no evidence is available for human carcinogenicity. PCBs are designated by EPA as class B carcinogens (Arthur D. Little, Inc. 1985).

## 8.3 RISK INDICATORS

### 8.3.1 Noncarcinogenic Risk

To quantitatively characterize the noncarcinogenic risk posed by contaminated groundwater associated with the Oil Landfarm WMA, a comparison is made between projected doses and references doses (RfDs) for the contaminants. EPA defines the RfD as the dose of a toxic agent (mg contaminant/kg body weight/day of exposure) which does not result in any adverse effects under conditions of chronic exposure of the general human population, including sensitive subgroups. RfDs are generally calculated by dividing the no-observed-adverse-effects-level (NOAEL) (derived from animal or human toxicity studies) by an uncertainty factor to insure protection of human health. The recommended safety factors are 10, 100, or 1000 depending on the type of data used to determine the NOAEL. RfDs are available for PCE and 1,1-DCE and are shown in Table 8.1.

Table 8.1. Oral reference doses and cancer potency factors

| Contaminant | Reference dose (mg/kg/day) | Cancer potency factor $(mg/kg/day)^{-1}$ |
|---|---|---|
| Tetrachloroethene | $1.0 \times 10^{-2}$ | $5.1 \times 10^{-2}$ |
| Trichloroethene | $a$ | $1.1 \times 10^{-2}$ |
| 1,1-Dichloroethene | $9.0 \times 10^{-3}$ | $5.8 \times 10^{-1}$ |

$^a$No value available.

*Source:* EPA 1986.

For noncarcinogens a measure of potential health risk is obtained by dividing the exposure dose (mg/kg/day) of a chemical by the RfD. Doses are calculated from concentrations by assuming ingestion of 2 L of water a day and a body weight of 70 kg. The RfD is considered safe for humans; if the ratio of the exposure dose to the RfD exceeds one, a potential health risk exists. If the site involves more than one contaminant, the hazard index is used to express the risk. The hazard index is defined as the sum of the ratios of dose to reference dose for each contaminant (EPA 1986). A hazard index greater than unity (one) represents a potential health risk. Because no RfD is available for TCE, it is excluded from the hazard index calculations.

## 8.3.2 Carcinogenic Risk

Carcinogenic risk to humans is measured in terms of excess cancer risk. Excess cancer risk is defined by EPA as the probability (at the 95% confidence level) that an individual will develop cancer after a 70-year exposure to the chemical. Excess means that the risk is in addition to the risk of cancers presented by other sources. The excess cancer risk is estimated using the EPA Cancer Potency Factor (CPF) $[(mg/kg/day)^{-1}]$, which is defined as the upper limit on the lifetime probability that the carcinogen will cause cancer at a dose of 1 mg/kg/day. For a single chemical the excess cancer risk is equal to the product of the CPF and the exposure dose. For a group of contaminants the total excess cancer risk is equal to the sum of the excess cancer risks for all contaminants. PCE, TCE, and 1,1-dichloroethene are considered carcinogens; CPFs for these compounds are shown in Table 8.1.

Although there is no single definition of an acceptable cancer risk, a $10^{-6}$ probability of developing cancer in a 70-year lifetime is often referred to as an "acceptable risk" level (Whipple, 1988). Examination of past EPA regulatory decrees indicates that risks in the range of $10^{-4}$ to $10^{-7}$ have proven to be acceptable levels of risk (Travis and Hattemer-Frey, 1988), depending on site-specific factors such as land use or availability of exposure pathways. This study assumes that risks lower than $10^{-4}$ are within the range of acceptability.

## 8.4 RESULTS OF RISK ASSESSMENT

### 8.4.1 Groundwater

#### 8.4.1.1 Introduction

Hazard indexes and excess cancer risks are calculated at two locations, one on the east edge of the WMA (location E) and one on the west edge of the WMA (location W) (Fig. 8.1). Location E exhibits the highest hazard indexes at any location outside the WMA, and location W shows the highest excess cancer risks for any location outside the WMA. Values of the risk indicators based on locations E and W are, therefore, worst-case estimates for noncarcinogenic and carcinogenic risk, respectively. As is obvious from inspection of the concentration contour plots presented in Sect. 7, risks fall off very rapidly with distance from the WMA, and both noncarcinogenic and carcinogenic risk are zero at less than 2000 ft downgradient (SW) from the site for all periods simulated.

#### 8.4.1.2 Noncarcinogenic risk

Figures 8.2 and 8.3 show hazard indexes as a function of time for the no-action alternative and for the capping and recovery well alternatives without continuous source terms for locations E and W, respectively. At location E the hazard indexes for all alternatives are above 1 in 1988. The hazard index for the no-action alternative remains slightly above 2 indefinitely because of the continuous source terms in the simulations. For the other alternatives the hazard indexes fall off rapidly and are below 1 by ten years. The recovery well alternatives are little more effective than capping in reducing noncarcinogenic risks.

At location W the hazard indexes for all alternatives are zero in 1988 and then rise as the plumes migrate outward toward this location. For the no-action alternative the hazard index rises above 1 at about 15 years and remains between 1 and 2 indefinitely. For the other alternatives hazard indexes are never above 1. At this location also the recovery well alternatives are little more effective than capping in reducing risks. Figures 8.4 and 8.5 show the hazard indexes for capping and the recovery well alternatives with source terms for locations E and W, respectively. In these figures the capping curve is identical to the no-action curve in the previous figures. At location E the capping hazard index remains slightly above 2 indefinitely. The hazard indexes for the other alternatives at this location fall rapidly after recovery begins in 1993 and are below 1 by 15 to 20 years. They then remain at constant levels indefinitely because of the continuous source terms in the simulations.

At location W all hazard indexes are zero in 1988. The capping hazard index rises above 1 by about 15 years and remains so indefinitely. The hazard index for the 600-ft recovery case rises only very slightly above 1 at about 15 years, and for the 200-ft case never rises above 1 at this location. At both locations the predicted noncarcinogenic risk levels for the alternatives under the continuous source-term scenario differ somewhat more than they did for the no source-term scenario, but only by factors of 2 to 3.

#### 8.4.1.3 Carcinogenic risk

Figures 8.6 and 8.7 show excess cancer risks as a function of time for the no-action alternative and the capping and recovery well alternatives without source terms for locations
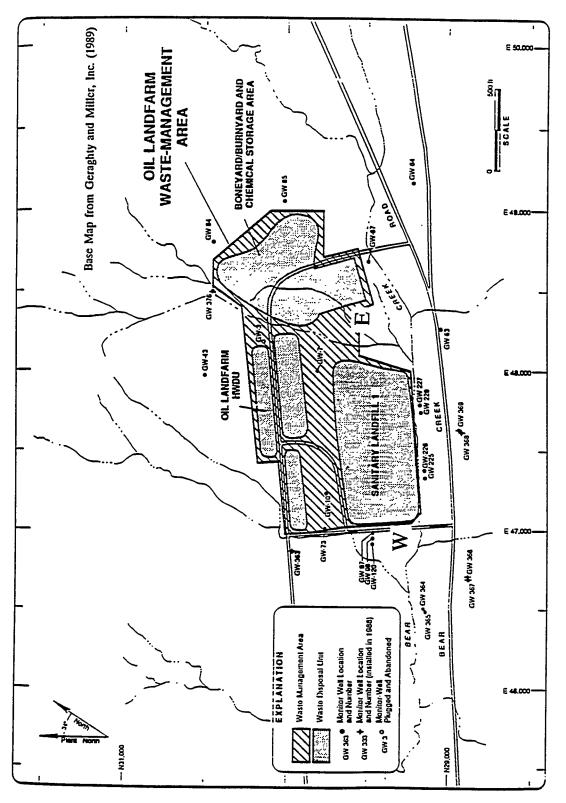
Fig. 8.1. Risk assessment locations.
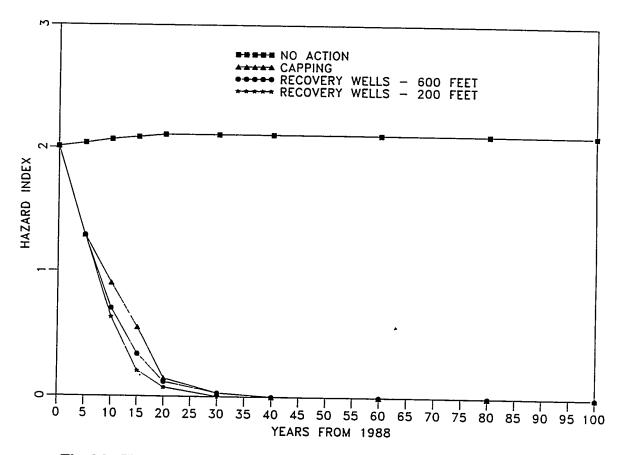
**Fig. 8.2. Hazard index at location E as a function of time.** No source terms in the capping and recovery simulations.
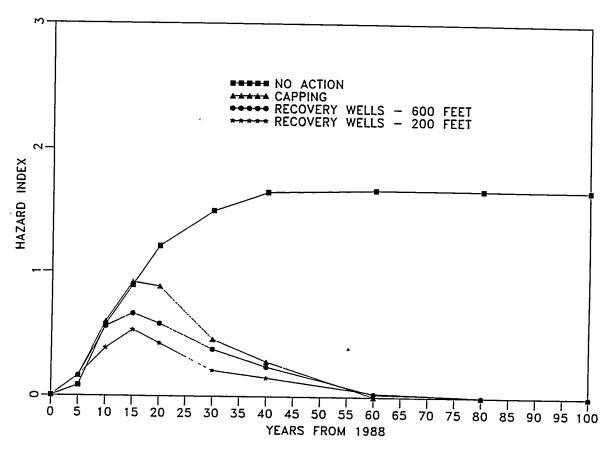
Fig. 8.3. Hazard index at location W as a function of time. No source terms in the capping and recovery simulations.
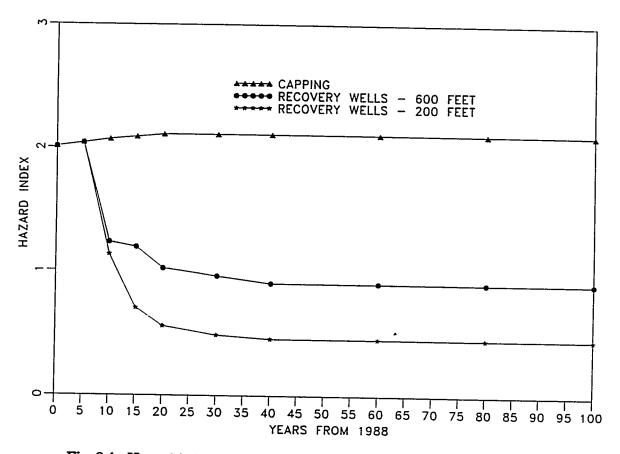
**Fig. 8.4. Hazard index at location E as a function of time.** Source terms run continuously throughout the capping and recovery simulations.
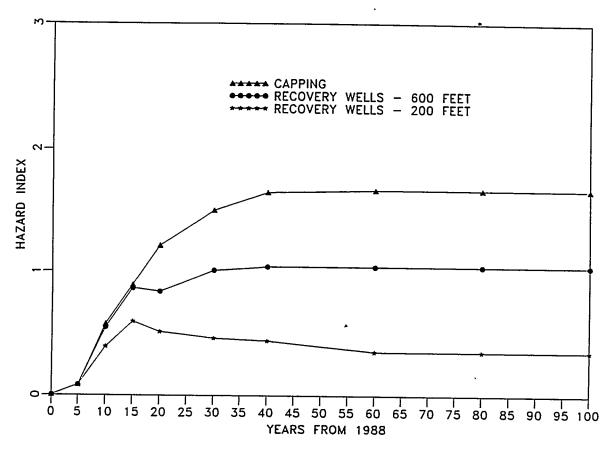
**Fig. 8.5. Hazard index at location W as a function of time.** Source terms run continuously throughout the capping and recovery simulations.
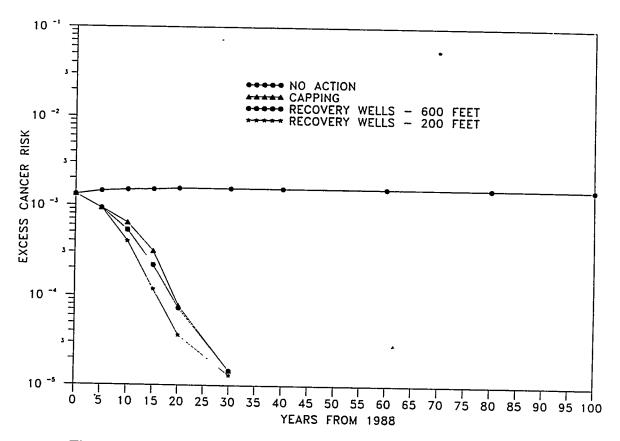
**Fig. 8.6. Excess cancer risk at location E as a function of time.** No source terms in the capping and recovery simulations.
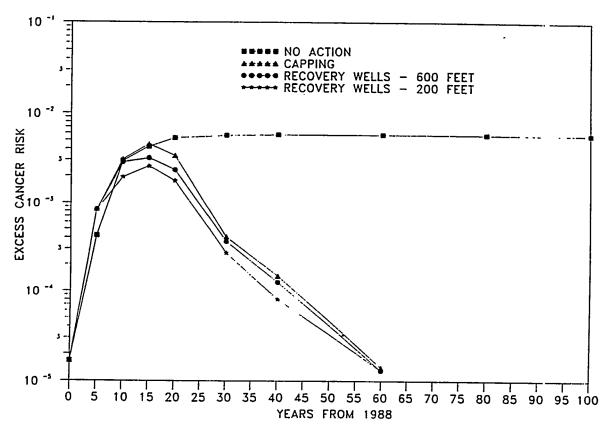
**Fig. 8.7. Excess cancer risk at location W as a function of time.** No source terms in the capping and recovery simulations.

E and W, respectively. At location E the no-action cancer risk is above $10^{-3}$ indefinitely. Excess cancer risks for the other alternatives fall rapidly and are below $10^{-4}$ by about 20 years. At location W excess cancer risks initially rise and reach levels well above $10^{-3}$ for all alternatives at 10 to 15 years. They remain above $10^{-3}$ for the no-action alternative indefinitely. For the other alternatives they begin to fall after 15 years and are below $10^{-4}$ by about 40 years. There is little difference between risk levels for the capping and the recovery well alternatives at either location.

Figures 8.8 and 8.9 show excess cancer risks as a function of time for the capping and recovery well alternatives with source terms at locations E and W, respectively. At both locations cancer risks for all alternatives are above $10^{-4}$ indefinitely for this scenario. The recovery well alternatives do not produce significant cancer risk reductions with respect to capping.

### 8.4.2 Surface Water

As discussed in Sect. 7, contaminant concentrations in surface water can be expected to be roughly an order of magnitude below those found in groundwater discharging into the surface water. The highest concentrations of VOCs for the no-action or capping with source-term alternatives in Bear Creek immediately south of the Oil Landfarm WMA would then be in the low 10s of $\mu g/L$. Concentrations of this order do not produce hazard indexes above 1 and give excess cancer risks less than only slightly above $10^{-4}$. For the other alternatives risks would be considerably lower. Consequently, surface waters do not pose significant health risks at this site.

### 8.4.3 Air

Potential health risks associated with the emission of VOC-contaminated air from the GWTF are evaluated by comparing predicted concentrations at selected locations with acceptable concentrations, based either on Cancer Potency Factors or Threshold Limit Values (Table 8.2). Predicted on-site concentrations at the time of startup of the treatment module (approximately 1997) (Table 7.4, Figs. 7.37–7.40) exceed acceptable values in the immediate vicinity of the treatment module only for 1,1-DCE, and are below allowable values over the area of the main plant site for all VOCs. Highest off-site concentrations occur near a residence in Country Club Estates southwest of the treatment modules; these concentrations are below acceptable values for all VOCs. Downwind (i.e., west and southwest of this location), concentrations fall off rapidly and are well below values of concern for public health (Fig. 7.41). The concentrations of contaminants emitted from the GWTF depend primarily on contaminant concentrations in the recovered water from the Bear Creek Burial Grounds, and the contributions from the Oil Landfarm are relatively minor.

Because peak concentrations of VOCs in groundwater migrate relatively slowly to the recovery well network at the Burial Grounds, concentrations of VOCs in the water input to the GWTF and, subsequently, in the air emitted from the facility can be expected to increase for perhaps the first 20 years of operation. Air concentrations over the area modeled are linearly dependent on the output of the treatment module (O'Donnell, personal comm., 1988). Consequently, air concentrations at the locations discussed above may increase by roughly an order of magnitude by about 20 years from today. If this were the case, locations near the west end of the Y-12 Plant would probably exhibit air concentrations slightly in
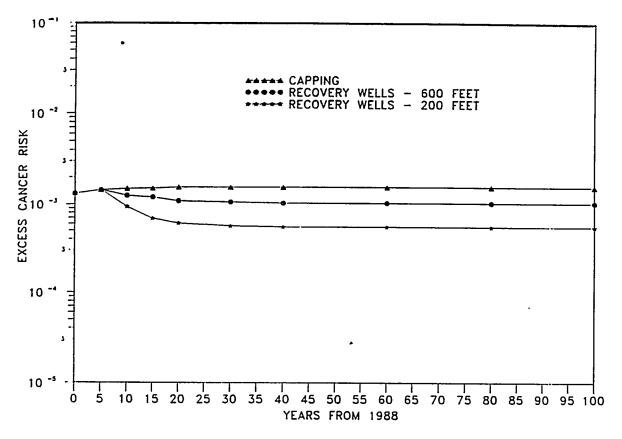
**Fig. 8.8. Excess cancer risk at location E as a function of time.** Source terms run continuously throughout the capping and recovery simulations.
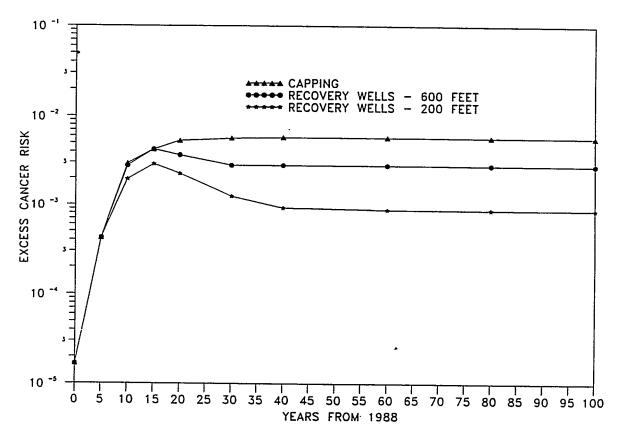
Fig. 8.9. Excess cancer risk at location W as a function of time. Source terms run continuously throughout the capping and recovery simulations.

excess of acceptable values. Maximum off-site concentrations in the vicinity of Country Club Estates would probably also exceed acceptable values for 1,1-DCE, but not for the other VOCs.

Table 8.2. Inhalation guidelines

| Compound | Guideline (pg/m$^3$) |
|---|---|
| Tetrachloroethene[a] | $2.0 \times 10^6$ |
| Trichloroethene[a] | $3.0 \times 10^5$ |
| trans-1,2-Dichloroethene[b] | $2.6 \times 10^8$ |
| 1,1-Dichloroethene[a] | $3.0 \times 10^3$ |

[a]Based on EPA's Cancer Assessment Group Inhalation Cancer Potency Factor at $10^{-6}$ cancer risk.

[b]Based on the American Conference of Governmental Industrial Hygienists' Threshold Limit Value.

# 9. RISK ASSESSMENT: ECOLOGICAL

## 9.1 PRESENT ECOLOGICAL CONDITIONS IN BEAR CREEK

### 9.1.1 Water Quality

Water quality in Bear Creek in the vicinity of the Oil Landfarm WMA is affected by inputs from upstream disposal sites, principally the contaminated groundwater plume at the S-3 Ponds site. Any inputs from the Oil Landfarm site and other nearby disposal sites discharge to tributaries NT3 and NT4 or directly to Bear Creek. A recent report on ecological conditions (including water quality) in Bear Creek, 1984 to 1987, noted little evidence of increases in the concentrations of any inorganic constituents in water or sediments in Bear Creek as a result of inputs from this area (Southworth et al., 1988).

The major pollutants that might enter Bear Creek as a result of leaching and erosion from the Oil Landfarm WMA are chlorinated organic solvents (tetrachloroethene, trichloroethene, dichloroethenes, vinyl chloride, trichloroethanes, and dichloroethanes) and hydrocarbon-derived oil containing polychlorinated biphenyls (PCBs). Transport pathways, and thus remedial control measures, asssociated with these two groups of organic compounds differ substantially. Chlorinated solvents are immiscible with and denser than water, but also have appreciable water solubility (Sect. 7.2.1). In landfill disposal, such materials sink into the soil below the water table and act as a continuing reservoir of contamination, slowly dissolving into the groundwater. Surface water contamination occurs when groundwater containing these solvents surfaces in streams. Once in the surface flow, these substances are rapidly removed by volatilization. Oil containing PCBs, however, is less dense than water and does not penetrate the water table but rather floats on it. Much of the transport to surface waters occurs as a result of the movement of oil-contaminated soils and debris (as well as floating oil films) following moderate to heavy rainfall.

No abrupt increase in concentrations of chlorinated solvents (hereafter referred to as VOCs), in Bear Creek is associated with the inputs of NT3 and NT4 or with passage of Bear Creek through the Oil Landfarm WMA (BCK 11.8–11.0) (Table 9.1). Although VOCs are present in groundwater at this site, the concentrations and extent of contamination are not high enough to result in detectable concentrations in Bear Creek. Maximum concentrations of individual organic compounds measured in Bear Creek or any of its tributaries, and EPA Water Quality Criteria (WQC) for the protection of freshwater aquatic biota from chronic toxicity are listed in Table 9.2. None of the organics are highly toxic, and maximum aqueous phase concentrations of all VOCs are well below WQC in Bear Creek and its tributaries. Dichloroethene and tetrachloroethene occur at concentrations closest to toxic levels.

Water quality monitoring does not consistently detect PCB contamination in Bear Creek; however, PCB measurements in sediments throughout the length of Bear Creek reveal an abrupt increase below NT7 similar to that observed for VOCs in water at the same location (Fig. 9.1). PCB concentrations in Bear Creek in the vicinity of the Oil Landfarm WMA are much lower. The concentrations of PCBs in sediments in the vicinity of BCK 9.9 would likely be associated with PCB concentrations approaching the 2 $\mu g/g$ Food and Drug Administration limit in fish if present in a larger stream. PCB contamination has been observed in biota in
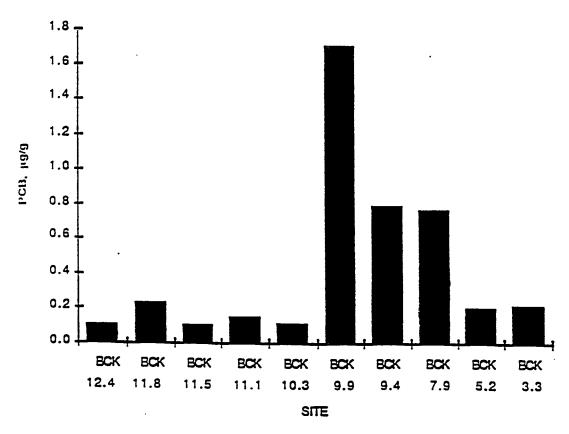
Fig. 9.1. PCBs in fine surface sediments ($\mu$g/g dry weight) at various sites in Bear Creek, showing input of PCBs from tributaries draining the Oil Landfarm.

lower Bear Creek; concentrations are significant but not alarming (Turner et al. 1989). Bear Creek is probably one of several important sources of PCB contamination to biota in lower Poplar Creek and nearby reaches of the Clinch River, but, the predominant source of this contamination is the Burial Grounds, not the sites in the Oil Landfarm area.

**Table 9.1. Summed concentrations ($\mu$g/L) of volatile organics in Bear Creek surface water at various sites**

| Site | Date | | | | | | |
|------|------|------|------|------|------|------|------|
| | 9/83[a] | 3/84[a] | 8/84[a] | 8/85[b] | 1/86[b] | 5/86[b] | 9/86[b] |
| BCK 12.39 | | | <10 | <10 | <10 | <10 | <10 |
| BCK 11.83 | | | <10 | <10 | <10 | <10 | <10 |
| BCK 11.49 | ND | <10 | ND | <10 | <10 | <10 | |
| BCK 11.09 | ND | ND | | | 10 | | <10 |
| BCK 10.32 | | ND | | <10 | <10 | | <10 |
| BCK 9.91 | 1291 | 201 | 225 | 147 | 874 | 210 | 210 |
| BCK 9.43 | 52 | 97 | 11 | 46 | 225 | 11 | 23 |
| BCK 9.40 | | | | 24 | 34 | <10 | 12 |
| BCK 7.87 | | | <10 | <10 | <10 | <10 | <10 |
| BCK 5.15 | | | | <10 | <10 | <10 | <10 |
| BCK 3.25 | | | | 65 | <10 | <10 | <10 |

[a]*Source:* Martin Marietta Energy Systems, Inc. (1985a).
[b]*Source:* Roy F. Weston Laboratory, Inc. (1986).
ND = no data.

Reaches of Bear Creek commonly go dry in summer as flow goes underground in losing reaches. The most extensive reach of intermittent stream is between BCK 11.64 and BCK 9.41; a smaller intermittent reach occurs above BCK 4.7 (Southworth et al. 1988). Periods of zero flow are most common near BCK 10.41, and occur less frequently at BCK 4.70 (Table 9.3). The north tributaries of Bear Creek above the SS5 spring, especially NT3, NT4, NT5, and NT6 are intermittent, and usually dry up during summer and early fall.

Table 9.2. Maximum concentrations of volatile organic compounds in Bear Creek
and tributaries compared with EPA water quality criteria

| Compound | Maximum concentration (mg/L) | | Water quality criterion[a] (mg/L) |
|---|---|---|---|
| | Bear Creek | Tributaries | |
| Tetrachloroethene | 0.102 | 0.35 | 0.84 |
| Trichloroethene | 0.082 | 0.29 | 21.9 |
| Dichloroethene[b] | 0.881 | 2.22 | 11.6[c] |
| Vinyl chloride | 0.092 | 0.85 | |
| Dichloroethane[b] | 0.092 | 0.55 | 20.0 |
| Trichloroethane | 0.078 | 0.22 | 2.4 |
| Chloroform | 0.050 | 0.028 | 1.24 |
| Toluene | 0.063 | 0.51 | 17.5[c] |

[a]EPA water quality criterion for the protection of freshwater biota from chronic toxicity.
[b]Summed maximum concentrations of isomers.
[c]Water quality criterion for acute toxicity; no chronic criterion as yet derived.


## 9.1.2 Fish Populations

Surveys of fish populations in Bear Creek from May 1984 to November 1987 (Table 9.4) indicate that populations in most of Bear Creek differ little from fish populations in local reference streams (Southworth et al. 1988). Three species of minnows dominate the populations; blacknose dace (*Rhinichthys atrulatus*), Tennessee dace (*Phoxinus tennesseenis*, formerly classified as mountain redbelly dace (*Phoxinus oreas*), and creek chub (*Semotilus atromaculatus*). Density and biomass are similar to those of unimpacted streams at all locations except the uppermost site, BCK 12.4. Unpublished data from late 1988 and summer 1989 indicate that robust fish populations (including large numbers of Tennesssee dace) are now found, at least part of the year, at BCK 12.4 (M. G. Ryon, personal communication).

In contrast to density and biomass, species richness (i.e., the number of species inhabiting a site) is slightly lower in Bear Creek above the weir at BCK 5.2 than it is in the larger reference stream (Mill Branch). This will remain unchanged as long as the weir continues to be a barrier to the upstream movement of fish. One important aspect of the fauna of Bear Creek above the weir is the distribution and abundance of the Tennessee dace (formerly called mountain redbelly dace, but recently reclassified). This species is deemed in need of management by the state of Tennessee, and its habitat is protected. It is found at every site above BCK 5.2, and constitutes a significant proportion of the fish communities at BCK 11.8, 11.1, 9.9, 9.4, 7.9 and, by now, possibly BCK 12.4. The fish communities at the two sites

(BCK 11.8 and 11.1) immediately downstream from the oil landfarm do not show any significant adverse effects when compared with sites farther downstream.

Table 9.3. Frequency and duration of zero flow periods in Bear Creek, 1984–87

| Site | Percentage of sampling dates with zero flow[a] | | | | No. of consecutive sampling dates of zero flow | | | |
|------|------|------|------|------|------|------|------|------|
|      | 1984 | 1985 | 1986 | 1987[b] | 1984 | 1985 | 1986 | 1987[b] |
| BCK 12.46 | 0[c] | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| BCK 11.64 | 0 | 0 | 0 | 4.0 | 0 | 0 | 0 | 1 |
| BCK 11.17 | 16.7 | 0 | 19.2 | 36.0 | 3 | 0 | 5 | 6 |
| BCK 10.41 | 54.2 | 38.5 | 76.9 | 68.0 | 74 | 2 | 6[d] | 14 |
| BCK 9.53 | 8.3 | 0 | 15.4 | 44.0 | 2 | 0 | 1 | 9 |
| BCK 9.41 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| BCK 7.87 | 0[c] | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| BCK 4.70 | NS[e] | 0 | 15.4 | 28.0 | NS | 0 | 3 | 6 |

[a]May through October only; $N = 24$ (1984), $N = 26$ (1985), $N = 27$ (1986), and $N = 25$ (1987).
[b]Actual values may be higher because no measurements were taken during October 17-31 when precipitation was low (total for that month was only 24% of normal).
[c]No flow measurements were taken prior to July 26, 1984.
[d]July 17 through August 22 and again from September 5 through October 9.
[e]NS = not sampled.

When water is present, the intermittent reach of Bear Creek between BCK 11.8 and BCK 9.4 contains fish populations typical of permanent reaches of the stream. Sites in this reach of stream represent habitat for reproduction and larval/juvenile development. Spawning and rearing habitat for the mountain redbelly dace and other species in the intermittent reach are comparable to those in the permanent reaches downstream, and likely play an important role in maintaining fish populations.

### 9.1.3 Benthic Macroinvertebrates

The benthic macroinvertebrate community in the vicinity of the Oil Landfarm is clearly affected by contamination relative to communities of such organisms found in uncontaminated local streams of similar size, but the cause of these ecological effects appears to be the input of contaminated groundwater from the S-3 Ponds plume in the headwaters of Bear Creek (Southworth et al. 1988). A clear pattern of steadily increasing density, species richness (mean number of taxa per sample), and diversity with increasing distance from the S-3 Ponds

Table 9.4. Total fish density (individuals/m²), total biomass (g/m²), and species richness for May 1984 to November 1987 in Bear Creek (BCK) and reference streams [Grassy Creek (GCK) and Mill Branch (MBK)]

| Site[a] | BCK 12.4 | BCK 11.8 | BCK 11.1 | BCK 9.9 | BCK 9.4 | BCK 7.9 | BCK 3.3 | GCK 2.4 | MBK 1.6 |
|---|---|---|---|---|---|---|---|---|---|
| **Sampling period:** | | | | | | | | | |
| **May–June 1984** | | | | | | | | | |
| Density | 0 | 0.08 | 0.73 | 0.86 | 0.85 | 1.11 | 1.72 | 1.24 | NS[b] |
| Biomass | 0 | 0.40 | 0.81 | 2.44 | 2.77 | 3.08 | 7.49 | 2.45 | NS |
| Richness | 0 | 2 | 1 | 3 | 5 | 5 | 9 | 5 | NS |
| **March–April 1985** | | | | | | | | | |
| Density | 0.10 | 3.85 | 2.50 | 1.88 | 1.07 | 4.01 | 1.67 | 1.59 | 2.64 |
| Biomass | 0.43 | 5.97 | 5.42 | 5.87 | 3.86 | 7.89 | 8.51 | 2.88 | 4.91 |
| Richness | 2 | 3 | 4 | 6 | 6 | 5 | 11 | 6 | 7 |
| **July–August 1985** | | | | | | | | | |
| Density | 0.02 | 3.76 | 0.02 | 2.28 | 6.65 | 8.56 | 7.19 | 3.90 | 4.26 |
| Richness | 1 | 3 | 1 | 4 | 6 | 5 | 9 | 6 | 8 |
| **November–December 1985** | | | | | | | | | |
| Density | 0.01 | 3.92 | 0.98 | 0.93 | 1.81 | 4.47 | 1.35 | 0.82 | 1.52 |
| Biomass | 0.01 | 9.83 | 4.03 | 2.41 | 3.58 | 6.95 | 3.63 | 2.03 | 2.73 |
| Richness | 1 | 3 | 4 | 4 | 6 | 6 | 11 | 4 | 7 |
| **March–April 1986** | | | | | | | | | |
| Density | 0.29 | 1.36 | 0.96 | 1.38 | 1.58 | 3.52 | 1.48 | 0.86 | 1.33 |
| Biomass | 1.04 | 2.35 | 1.82 | 3.14 | 4.47 | 6.59 | 6.82 | 1.92 | 2.28 |
| Richness | 2 | 3 | 4 | 4 | 6 | 6 | 10 | 4 | 5 |

Table 9.4 (continued)

| Site[a] | BCK 12.4 | BCK 11.8 | BCK 11.1 | BCK 9.9 | BCK 9.4 | BCK 7.9 | BCK 3.3 | GCK 2.4 | MBK 1.6 |
|---|---|---|---|---|---|---|---|---|---|
| **Sampling period:** | | | | | | | | | |
| **November–January 1986/87** | | | | | | | | | |
| Density | 0 | 3.43 | 1.54 | 1.58 | 3.92 | 5.70 | 0.91 | 1.12 | 2.21 |
| Biomass | 0 | 8.40 | 4.21 | 2.20 | 5.66 | 6.30 | 2.12 | 2.16 | 2.11 |
| Richness | 0 | 4 | 4 | 5 | 6 | 6 | 10 | 4 | 6 |
| **March–April 1987** | | | | | | | | | |
| Density | 0.17 | 1.80 | 0.93 | 2.62 | 1.66 | 4.33 | 1.11 | 0.76 | 1.66 |
| Biomass | 0.90 | 6.21 | 2.75 | 8.80 | 3.52 | 6.55 | 2.72 | 2.07 | 1.60 |
| Richness | 3 | 4 | 4 | 5 | 6 | 6 | 11 | 4 | 7 |
| **October–November 1987** | | | | | | | | | |
| Density | 0 | 1.83 | 1.26 | 2.03 | 5.97 | 3.46 | 1.44 | 1.16 | 2.30 |
| Biomass | 0 | 2.61 | 0.74 | 4.29 | 9.09 | 5.08 | 2.86 | 2.96 | 2.58 |
| Richness | 1 | 3 | 4 | 4 | 5 | 5 | 10 | 3 | 8 |

[a]Data from other sites (BCK 10.3, BCK 10.0, BCK 5.2, and GCK 1.4) were evaluated but not tabulated because those collections did not include all dates).

[b]NS = not sampled.

is evident in the macroinvertebrate fauna. There is no abrupt change in this pattern associated with inputs from NT3 and NT4, but, a substantial improvement in the ecological "health" of the benthic community is apparent downstream from the input of the SS5 spring at BCK 9.41.

The nature of the composition of the benthic fauna in most of Bear Creek suggests of toxic effects resulting from contamination by metals. Mayflies, which are very sensitive to contamination with heavy metals, are rare or absent throughout most of Bear Creek, but caddisflies and stoneflies, pollution-sensitive taxa that are more tolerant of metal contamination, are moderately abundant at sites in the vicinity of the Burial Grounds and downstream. The stonefly Amphineura, a member of a family known to be particularly tolerant of acid conditions and elevated metals concentrations, is comon at all sites except BCK 12.4.

We find no evidence that the Oil Landfarm area has any effect on the benthic macroinvertebrate fauna of Bear Creek. To exhibit visible effects of contamination deriving from the Oil Landfarm area, Bear Creek would have to eliminate or reduce population densities of pollution tolerant species, but these species are already absent upstream of the site due to pollution from the S-3 Ponds site.

### 9.1.4 Toxicity Studies

Toxicity tests using fathead minnows were conducted on Bear Creek water eight times between 1984 and 1988. Toxicity was generally not observed at BCK 11.8, BCK 11.1, and BCK 10.8 on those tests, but was usually observed upstream at BCK 12.4 (Southworth et al. 1988). There is no indication of an increase in toxicity of Bear Creek water in the vicinity of the Oil Landfarm in any of the eight tests. Toxicity tests were also run using the invertebrate Ceriodaphnia in March 1988 (White et al. 1989; Southworth et al. 1988). Toxicity was observed in Bear Creek water from BCK 11.8 but not at BCK 10.8 (sites receiving inputs from tributaries draining the Oil Landfarm). Water at an upstream site (BCK 12.4 near the S-3 Ponds) was clearly toxic to Ceriodaphnia, at a greater dilution than caused toxicity at BCK 11.8. Water in Bear Creek near the Oil Landfarm thus does appear to be toxic in toxicity testing, but, the source of the toxicity is the S-3 contaminant plume upstream.

## 9.2 ECOLOGICAL IMPACTS OF REMEDIAL ALTERNATIVES

### 9.2.1 No-Action Option

Under the no-action option, rainfall would continue to percolate through the Oil Landfarm WMA and introduce solutes to the groundwater. As was the case at the Bear Creek Burial Grounds WMA (Walter et al. 1990), the VOC concentrations in the contaminated groundwater plume at the Oil Landfarm WMA are predicted to increase over time at sites within several hundred meters from the source. Concentrations of VOCs in surface waters are not explicitly predicted at this site because of insufficient data needed for calibrating calculations. However, surface water calculations applied to the Bear Creek Burial Grounds generally predict surface water VOC concentrations to be roughly a factor of 10 less than concentrations in groundwater underlying the stream site. Maximum predicted

concentrations of PCE, TCE, and 1,1-DCE in groundwater underlying Bear Creek approximate 200 $\mu g/L$ (see contour plots in Sect. 7), implying surface water concentrations of roughly 10–50 $\mu g/L$. Such concentrations are well below the EPA Water Quality Criteria (WQC) for the protection of aquatic life (Table 9.2) of 840, 21,900, and 11,600 $\mu g/L$ for PCE, TCE, and DCE, respectively. Even the predicted concentrations in groundwater fall well below WQC. Continued expansion of the VOC-contaminated groundwater plumes at the Oil Landfarm WMA (with eventual discharge of VOCs to Bear Creek surface flow) is therefore unlikely to result in adverse effects on aquatic biota.

PCE and TCE are degraded by microbial action at a very slow rate in anaerobic groundwaters, and the presence of degradation products (dichloroethene and vinyl chloride) in groundwater at the Bear Creek Burial Grounds and the Oil Landfarm WMA indicates that some biodegradation has occurred. Because estimates of biodegradation rates of VOCs in groundwater are very low but highly uncertain, all VOCs are assumed to not be biodegraded in the transport model. Modeling results are thus conservative with respect to VOC transport; maximum concentrations of VOCs predicted far into the future probably may not be attained due to their slow biodegadation.

## 9.2.2 Cap and Leachate Control Options

Our analysis predicts that the cap and leachate control option, without continued contaminant input from possible NAPL sources, will effectively reduce the spread of groundwater contaminants and shorten the duration of contamination where groundwater infiltrates Bear Creek (Sect. 7). With continued contaminant input, groundwater contamination is expected to resemble no-action results. However, since the present and predicted future concentrations of contaminants are not high enough to pose an ecological threat even for no-action, capping should result in no discernible ecological benefit. Increased runoff from the impermeable cap and decreased infiltration of rainwater will result in slightly increased flow variability in Bear Creek in the vicinity of the capped area. Siltation effects are possible, but can be controlled through good management practices.

## 9.2.3 Cap and Groundwater Withdrawal and Treatment

### 9.2.3.1 Effects on water quality

Inputs of VOCs to Bear Creek are predicted to increase in the future despite the use of a network of downgradient wells to withdraw groundwater and impede further migration of the contaminant plume. However, predicted concentrations are lower than those expected with either the no action or capping-only alternatives, and thus are not expected to have any adverse ecological effects. While present plans call for discharge of treated groundwater to East Fork Poplar Creek, the discharge of treated groundwater to Bear Creek should present little hazard to aquatic life, because VOCs are readily removed by air-stripping (which does not add any potentially toxic materials to the water). Removal of PCBs and residual hydrocarbons by carbon filtration similarly should eliminate these contaminants without adding toxicity.

## 9.2.3.2 Ecological consequences of groundwater withdrawals

If treated groundwater removed from the well network is not returned to local tributaries and Bear Creek, the likely result would be an increased frequency of no-flow conditions in NT3 and NT4 and the intermittent section of Bear Creek between BCK 11.90 (upstream from the mouth of NT3) and BCK 9.41 (where inputs from the SS5 spring restore permanent flow to Bear Creek). The water table is predicted to be lowered several feet below the present surface of NT3 and NT4 by the network of groundwater withdrawal wells, which would result in these streams being without surface flow most of the time. Discharging treated groundwater to this stream above the dewatered reach would be unreasonable because that water would infiltrate into the subsurface and migrate towards the well network, reducing its efficacy for containing the contaminant plume. Thus, both Bear Creek and the tributaries will undoubtedly become even more ephemeral in character under these circumstances.

The removal of 12 gpm (0.75 L/s) of groundwater from the Oil Landfarm WMA watershed represents about 6% of the 190 gpm (12.0 L/s) mean annual flow in the reach between BCK 11.64 and BCK 10.41 observed from 1985 to 1987 (Southworth et al. 1988). No-flow conditions occur regularly in this reach during the summer (May through October). In the summers of 1984, 1985, and 1986, BCK 11.17 was dry on 16.7, 0, and 19.2% of the days when weekly-flow measurements were made; during the third consecutive year of below normal precipitation, 1987, this reach was dry on 36% of the sampling dates (Table 9.3). Flow was absent in Bear Creek at BCK 10.41 on 38.5 to 76.9% of sampling dates in each summer over this same period. The effect of withdrawing groundwater from the Oil Landfarm WMA watershed and not discharging it to Bear Creek would adversely affect a 2-km reach of Bear Creek which would be dry somewhat more often during summer months than at present. The normally intermittent zone of Bear Creek is rapidly recolonized by fish (and to a lesser extent by benthic macroinvertebrates) during periods of consistent flow. These reaches of temporary habitat provide sites for spawning and also critical habitat for larval and juvenile fishes. If the periods of no flow and low flow become more extensive, the ability of intermittent reaches of the stream to play an important role in fish reproduction will be adversely affected. Similarly, the ability of this reach to support benthic invertebrates, and thus food for fish, would also be diminished.

The toxic effects of the S-3 contaminant plume in Bear Creek are attenuated by the dilution of the S-3 inputs as additional ground and surface water enters Bear Creek at downstream sites. Results of toxicity tests on water from upper Bear Creek suggest that toxicity (to Ceriodaphnia) may be observable as far downstream as the Burial Grounds (White et al. 1989). Removing dilution water may, therefore, act to increase the severity and extent of toxicological effects. In the reach of stream affected by withdrawals, ecological effects of diminished dilution would be manifest as rather subtle changes in community structure in the benthic invertebrate fauna, acting primarily to limit the presence and abundance of sensitive species such as mayflies, stoneflies, and caddisflies. Fish populations would not be expected to be affected. Conversely, discharge of treated groundwater to Bear Creek upstream from BCK 12.4 would act to dilute toxic constituents in this reach of stream where flow is small and most inputs from the S-3 plume occcur. Such a discharge would act to accelerate the ongoing process of ecological recovery in Bear Creek.

## 9.2.4 Effects of Remedial Actions on East Fork Poplar Creek

If treated groundwater from the Oil Landfarm WMA is discharged to upper East Fork Poplar Creek, it will be required to be nontoxic at a dilution level well in excess of the in-stream waste concentration. Such a discharge is thus unlikely to have adverse ecological effects. Treatment of the groundwater is not likely to require the addition of chemicals, and reduction of VOCs to nontoxic levels should be readily achieveable. Thus, the discharge of nontoxic treated water to upper East Fork Poplar Creek may improve water quality in the receiving stream; however the effect of the 17,250 gal/day discharge on the 7.4 million gal/day flow of upper East Fork Poplar Creek is likely to be negligible.

## 9.2.5 PCB Transport and Bioaccumulation

Much of the surface soil in the Oil Landfarm disposal plots is contaminated with PCBs. Although concentrations exceed 25 ppm only in a few places, general low level contamination (5–10 ppm) is characteristic of the site (Herbes 1989). Erosion of PCB-contaminated soil from the Oil Landfarm could act as a source of contamination to Bear Creek; sediment data suggest that little movement of PCBs to Bear Creek is occuring at this site (Turner et al. 1989). Closure and capping of the Oil Landfarm WMA removed contaminated soils containing >25 ppm PCBs, and will cover the remaining PCB-contaminated soil with an engineered cap, ensuring that no further potential for erosion of PCB-contaminated soil exists.

Groudwater contamination by PCBs has not been observed at this site (Turner et al. 1989), and this pathway is generally not significant for hydrophobic substances such as these. Thus, neither capping nor capping with groundwater withdrawal would have any effect on the movement of PCBs via groundwater. Transport of PCBs through the subsurface as components of a separate oil phase floating on the surface of a shallow or perched water table, as is the case at the Burial Grounds WMA, has not been noted at this site and would not be expected given the nature of the disposal method.

## 9.2.6 Other Environmental Effects

Substantial secondary environmental effects are associated with the capping of the Y-12 WMAs. Material for construction of the clay cap must be mined from a large borrow area on Chestnut Ridge. Construction of this source requires the destruction of roughly 40 acres of hardwood forest, to be replaced with a large area of bare subsoil having little value as wildlife habitat. Additional acreage of woodland and field habitat must be destroyed to construct the roadway for hauling soil from the borrow area to the sites being capped. Construction activities and large-scale deforestation within the Bear Creek watershed could be expected to adversely affect portions of Bear Creek and tributaries through siltation. This has in fact occured in reaches downstream from the West Borrow Area. Such effects would be expected to be temporary and amenable to remediation.

## 9.3 SUMMARY AND CONCLUSIONS

There is liitle to indicate that contaminant transport from this site has had an ecological effect on Bear Creek, where ecological conditions are dominated by the effects of inputs from

the S-3 groundwater plume. Present and predicted future concentrations of contaminants in Bear Creek as a result of contributions from the Oil Landfarm WMA and other nearby disposal sites are not high enough to cause ecological harm under any of the proposed scenarios for remedial action. Placing an impermeable cap on the site will have little ecological effect on Bear Creek. Rapid runoff of precipitation from the cap will result in slightly higher variability in stream flows below the Oil Landfarm WMA than would be the case without the cap, but proper and effective erosion control measures should minimize the likelihood of siltation effects. Adverse ecological effects in Bear Creek are likely to be associated with the proposed removal and treatment of contaminated groundwater, with subsequent discharge to East Fork Poplar Creek. While such effects are not likely to be large, they could represent a setback in the continuing ecological recovery of Bear Creek. Groundwater withdrawals will result in flow reductions and increased intermittent character in a 2.5-km reach of Bear Creek downstream from the Oil Landfarm WMA. Slight decreases in aquatic habitat and stream productivity may occur. Because the toxicity of contaminants from the S-3 plume is reduced at downstream sites by inputs of less contaminated groundwater, reducing groundwater inputs may slightly exacerbate toxicity problems. Conversely, discharging treated groundwater to the headwaters of Bear Creek could act to ameliorate toxic groundwater inputs near the S-3 Ponds and would eliminate habitat loss caused by flow reduction.

All options associated with capping carry the ecological cost of destruction of terrestrial habitat and increased erosion and siltation. However, since construction of the borrow area and haul road have been completed, further consideration of these costs is not germane to the evaluation of alternatives. Thus, the options having the least ecological impact (given that terrestrial effects associated with the borrow area have already occurred) on Bear Creek are the no-action alternative and the cap and leachate control option.

Contaminant transport from the Oil Landfarm WMA presently has little or no ecological impact on Bear Creek. Ecological effects (other than deforestation) associated with various remedial measures are likely to be relatively small, but generally are not offset by any ecological benefits. Ecological factors are unlikely to be a major determinant of the optimum remedy at this site.

# 10. CONCLUSIONS AND RECOMMENDATIONS

## 10.1 SUMMARY

The Oil Landfarm Waste Management Area (WMA) is located in Bear Creek Valley about 1 mile southwest of the Y-12 Plant on the U.S. Department of Energy (DOE) Oak Ridge Reservation. From 1943 until 1982 several types of solid and liquid wastes were deposited in the five disposal areas which constitute the Oil Landfarm WMA. The disposal areas are: the Oil Landfarm disposal plots, the Boneyard, the Burnyard, the Chemical Storage Area, and the Sanitary Landfill. The Oil Landfarm disposal plots were used for the biological degradation of oily wastes from 1973 until 1982. The Boneyard was active from 1943 to 1970 and received a great variety of wastes for burning or burial, including organics, metals, acids, and debris. The Burnyard is located beneath the present Chemical Storage Area. It operated from 1943 to 1968 and consisted of unlined trenches in which various wastes from plant operations were ignited with solvents or oils and burned. The Chemical Storage Area operated from 1975 to 1981 for the disposal of wastes which posed safety hazards (e.g., reactive, corrosive, and explosive chemicals). Finally, the Sanitary Landfill was used from 1968 to 1980 for the burial of solid wastes. Although the Sanitary Landfill was supposed to receive only nonhazardous wastes, it may contain toxic chemicals and contaminated materials.

Since 1982 the Y-12 Plant has sampled groundwater, surface water, soils, and sediments in Bear Creek Valley. Data from this sampling program show that at the Oil Landfarm WMA, groundwater is the most seriously contaminated medium. The chief contaminants of groundwater are the volatile organic compounds (VOCs), and concentrations of several VOCs are in the range of 100s to 1000s of $\mu g/L$ in shallow wells adjacent to waste disposal areas. Elevated levels of some metals, gross alpha, and gross beta occur in isolated wells. Groundwater contamination currently extends only a few hundred feet from the site. Soils in the immediate vicinity of the disposal areas contain elevated levels of organic compounds, metals, and polychlorinated biphenyls (PCBs). Stream sediments at the site contain only slightly elevated levels of some metals. Surface waters at the Oil Landfarm WMA contain elevated levels of uranium and trace amounts of VOCs.

A network of monitoring wells is in place at the Oil Landfarm WMA, and groundwater assessment monitoring as prescribed under the Resource Conservation and Recovery Act (RCRA) is performed on a quarterly basis. In addition several aquifer tests have been performed in the valley, providing much information on the hydrogeological system. The Y-12 Plant has also undertaken various pollution containment measures. The Sanitary Landfill was capped with clay and topsoil and revegetated in 1983. In 1989 large volumes of contaminated soils were excavated from the Oil Landfarm disposal plots, and the plots and the Chemical Storage Area have been covered with multilayer, engineered caps. Capping of the Boneyard is under way. Additional remedial actions are currently being evaluated as part of the Closure and Postclosure Activities (CAPCA) Project.

This study assesses the risks to human health and the environment posed by the Oil Landfarm WMA under three remedial scenarios:

1. take no action (reference case);

2. place impermeable caps over the disposal areas to reduce leachate production and contaminant input to the aquifer; and

3. in addition to the measures described in (2), construct a network of recovery wells to extract contaminated groundwater for treatment in an air-stripping facility.

Two cases are considered for the recovery well option: wells pumping over only the upper 200 ft of the aquifer and wells pumping over 600 ft of aquifer. In addition, we attempt to estimate the effects of possible long-term contaminant sources on the capping and recovery alternatives.

Based on their potential for harm to the public health, three contaminants are chosen for risk assessment: tetrachloroethene (PCE), trichloroethene (TCE), and 1,1-dichloroethene (1,1-DCE). The potential for migration of these contaminants via transport in groundwater, surface water, air, and movement with soils and sediments is evaluated using computer modeling and qualitative arguments. The results of this analysis show that groundwater is the most important migration pathway for the contaminants. Transport in surface waters, air, and movement via soils and sediments are likely to be unimportant compared to transport in groundwater.

Because the VOCs have low water solubilities and are denser than water, there is a good possibility that a portion of the VOCs disposed of at the Oil Landfarm WMA are now in the saturated zone as nonaqueous phase liquids (NAPLs). If this is the case, the NAPLs can be expected to act as long-term sources of contaminants to the aquifer in spite of capping. Because it is not possible to predict the actual effectiveness of capping, two opposite capping scenarios are modeled: (1) fully effective caps, which eliminate sources of contaminants to the aquifer and (2) completely ineffective caps, which do not decrease contaminant input to the aquifer. In the first case model, sources (source terms) are turned off during capping and recovery simulations. In the second case model, sources run throughout the simulations.

Because of the hydrogeological properties of the site, computer modeling predicts that the maximum extent of migration of the contaminants in groundwater under all scenarios is <2000 ft downgradient (SW) from the southwest corner of the WMA. Migration is limited largely because the plumes are predicted eventually to intersect Bear Creek and discharge contaminants into surface water. Capping does not significantly affect the extent of migration, and the 600-ft recovery well option has relatively little effect on this aspect of the contaminant plumes. The 200-ft recovery well option is only slightly more effective in limiting migration. Whether or not sources run throughout the simulations does not significantly affect the extent of plume migration.

Although the extent of migration is predicted to be limited, high concentrations of VOCs persist at the site for long times under all scenarios (i.e., about 40 years from today or until 2030). Because there are no data from which to constrain source-term durations, no-action and capping-with-sources and recovery-with-sources computer simulations use constant source terms throughout the full 100 years of simulations as worst-case scenarios. Although this procedure almost certainly overestimates the persistence of the predicted plumes, plume areas and peak concentrations eventually reach steady-state conditions because of the discharge of

contaminants to Bear Creek. Without sources in the simulations, plume areas and peak concentrations produced by the capping simulations are significantly smaller than those from the no-action simulations (the upgradient boundaries of the plumes migrate away from their original sources), and the simulated plumes persist for 30 to 60 years. With source terms, the results of capping simulations are essentially the same as the no-action results, and the plumes persist indefinitely.

The presence or absence of sources in the recovery simulations does not change the relative efficacy of recovery with respect to capping or of recovery over the two depths considered. Either with or without sources, recovery in the 600-ft case is little more effective than capping in reducing contaminant concentrations and plume areas. Without sources, the 600-ft recovery plumes persist from about 30 to 60 years; with sources, of course, they persist indefinitely. The 200-ft case, either with or without sources, is somewhat more efficient, and both peak concentrations and plume areas are reduced by about a factor of 2 with respect to the capping and 600 ft results at given times. Without sources, the plumes dissipate in about 25 to 40 years; with sources, they persist indefinitely. The recovery well networks are not very efficient because of restrictions on the placement of wells imposed by the proximity of Bear Creek, the cavity-ridden Maynardville Formation and limitations on pumping rates in other formations. In addition, relatively slow transport velocities, especially for PCE, extend cleanup times.

A quantitative assessment of the risks to human health posed by groundwater was performed using predicted contaminant concentrations. Because of the limited extent of migration of the contaminant plumes under all scenarios, significant health risks occur only in the vicinity of the site. However, for all alternatives, health risks at the site are above acceptable levels for longer than 40 years. For no-source scenarios, capping reduces health risks significantly with respect to the no-action alternative, but the recovery well network scenarios are little more effective than capping in reducing health risks. For scenarios with sources, health risks associated with capping are the same as for no action, but the recovery well scenarios are still not much more effective than capping in reducing risks.

As alluded to earlier, modeling predicts that VOC-contaminated groundwater eventually discharges to Bear Creek along the reach immediately south and southwest of the Oil Landfarm WMA. Because of dilution and loss of contaminants to air by volatilization, concentrations of contaminants in Bear Creek can be expected to be much lower than those in groundwater and to fall off rapidly downstream from the Oil Landfarm WMA. Because there are no data from which to calibrate them, calculations of future concentrations in Bear Creek were not attempted for this site. However, based on groundwater concentrations and previous experience, we would expect peak no-action concentrations in Bear Creek immediately south of the WMA to be of the order of 10s of $\mu g/L$, and peak concentrations for the other alternatives to be even less.

Under the recovery well scenarios, groundwater extracted from the Oil Landfarm WMA site is to be transported to the Bear Creek Burial Grounds where it will be treated together with groundwater from that site in an air-stripping facility, the Groundwater Treatment Facility (GWTF). Operation of the GWTF would result in discharge of contaminants to the atmosphere. Air modeling predicts that concentrations of contaminants in air at the time of GWTF startup (1997) as a result of the combined treatment of water from the Burial Grounds and from the Oil Landfarm WMA will be above acceptable levels only in the

immediate vicinity of the facility. The Oil Landfarm WMA's contribution to air contamination is almost negligible compared to that of the Burial Grounds.

Large volumes of PCB- and VOC-contaminated soil have already been excavated from the Oil Landfarm disposal plots. Emplacement of caps over the disposal areas should essentially eliminate erosion of soils and subsequent transport on sediments as pathways for contaminant migration. The recovery well option would have little additional effect on migration via these pathways.

There is little to indicate that contaminant transport from this site has had an ecological impact in Bear Creek, where ecological conditions are dominated by the effects of inputs from the S-3 groundwater plume. Present and predicted future concentrations of contaminants in Bear Creek as a result of contributions from the Oil Landfarm WMA and - other nearby disposal sites are not high enough to cause ecological harm.

Adverse ecological effects of the proposed remedial actions are likely to occur in Bear Creek. While such effects are not likely to be large, they could represent a setback in the continuing ecological recovery of Bear Creek. Groundwater withdrawals will result in flow reductions and increased intermittent character in a 2.5-km reach downstream from the Oil Landfarm WMA. Slight decreases in aquatic habitat and stream productivity are likely. Because the toxicity of contaminants from the S-3 plume is reduced at downstream sites by inputs of less contaminated groundwater, reducing groundwater inputs may slightly exacerbate toxicity problems. Conversely, discharging treated groundwater to the headwaters of Bear Creek could act to ameliorate toxic groundwater inputs near the S-3 pond site.

Estimated total costs for capping the disposal areas (not including the Sanitary Landfill) of the Oil Landfarm WMA are $8 million. Operating and maintenance costs for the cap are expected to be minimal. No cost estimate has been made for the specific recovery well and treatment scenarios considered in this study. However, based on other studies, capital costs are expected to be about $10 to $20 million, in addition to capping costs. Expected operation and maintenance costs are approximately $2 million/year.

## 10.2 CONCLUSIONS AND RECOMMENDATIONS

Our analysis of the alternatives just described leads to the following conclusions and recommendations:

1. Health risks associated with the Oil Landfarm WMA are confined to the immediate vicinity of the site.

2. Health risks under all alternatives can be expected to remain at unacceptable levels for more than 40 years at locations adjacent to the WMA.

3. The effect capping the disposal areas will have on contaminant input to groundwater is uncertain. If a significant portion of the contaminants at the site exist as NAPLs in the saturated zone, capping may not be very effective in reducing contaminant sources and ameliorating risks associated with groundwater at the site.

4. Recovery of contaminated groundwater with extraction wells is predicted to be little more effective than capping in reducing risk levels at this site. This conclusion is independent of whether or not long-term NAPL sources are present.

5. Several types of data are essential to more realistically evaluate the efficacy of the capping and recovery well scenarios. The locations and natures of the VOC sources need to be better characterized. The areal and vertical distributions of contaminants in groundwater need to be better understood to more fully evaluate recovery well scenarios, especially the depth over which contaminants should be recovered. Finally, more accurate data on the hydrologic properties of the formations at the site (e.g., sustainable yield per well and hydraulic conductivity as a function of depth), would be necessary for designing optimum recovery well networks or evaluating other alternatives.

6. Because exposure of the general public to contaminants from this site is unlikely in the foreseeable future, and because traditional groundwater recovery techniques are predicted to be relatively ineffective in reducing risk levels, we recommend that consideration be given to alternative remedial technologies (e.g., *in situ* biotreatment, steam displacement, and surfactants) rather than proceeding immediately with groundwater recovery and treatment. Realistically evaluating such alternative measures would require the same types of data outlined in (5) above.

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# Appendix

# GROUNDWATER MODELING

# A.1 INTRODUCTION

## A.1.1 Description of the Model

Contaminant transport in groundwater is simulated using the United States Geological Survey Two-Dimensional Solute Transport Method of Characteristics Model (MOC) (Konikow and Bredehoeft 1978). The model solves two simultaneous partial differential equations: the groundwater flow equation, which describes the distribution of hydraulic heads in the aquifer, and the solute transport equation, which gives the distribution of solute concentrations in the aquifer. The model is based on a rectangular, block-centered, finite-difference grid which corresponds to the surface area of the aquifer. In this study, flow is treated as steady-state. In addition to convective transport, hydrodynamic dispersion and adsorption are simulated. Transformation processes are not treated. Several aquifer and contaminant properties must be estimated and input to the model, and the model must be calibrated by matching current hydraulic head and contaminant distributions by model output. These aspects of modeling are discussed in detail below. The model output includes maps of hydraulic heads and concentrations of dissolved contaminants at finite-difference nodes at given times and concentration vs time data at selected locations.

## A.1.2 Modeling Assumptions

Three assumptions, which are germane to the Oil Landfarm WMA simulations, are inherent in the MOC model: (1) the aquifer is in a granular porous medium; (2) aquifer properties and contaminant distributions are uniform with depth; and (3) contaminants are transported as a single aqueous phase. The reasons that each of these assumptions may be questionable for the Oil Landfarm WMA are discussed below. Ideally, making these assumptions could be avoided by using more complicated models; however, in practice there are currently insufficient data to justify doing so for the purpose of risk assessment. The model limitations do not seriously affect the usefulness of the results; they should, however, be kept in mind when interpreting them.

The first assumption, that the permeability of the aquifer is produced by intergranular porosity, is questionable for the Oil Landfarm WMA because, as discussed in Sect. 2, in most of the formations of the Conasauga Group the permeability is created chiefly by fractures and, in the case of the Maynardville Limestone, partly by cavities. Although using porous media models for fractured media is common practice, there are potentially significant differences in the behavior of groundwater flow and contaminant transport for the two types of media (see, for example, Huyakorn, White, and Wadsworth 1987; Sudicky and Frind 1984). These differences are difficult to assess qualitatively, and there is at present no consensus on the conditions for which the use of porous media models for transport in fractured media is justified. Results of aquifer tests conducted in the Nolichucky in Bear Creek Valley (Lozier, Spiers and Pearson 1987) do, however, suggest the steady-state response of the aquifer can be adequately modeled with a porous media model such as MOC. Although some fractured media effects were observed in these tests, drawdown curves for most wells corresponded to the classic porous media pattern.

Use of a porous media model for a cavity-ridden formation such as the Maynardville involves more uncertainty. Hydraulic and physical properties and, consequently, flow paths and contaminant transport in such media can be quite erratic. Fortunately, because of the

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geology and topography of the site, flow paths are likely to be constrained to lie along the trend of the valley floor, giving confidence in the overall transport direction. However, the scale of this study is probably not large enough to permit hydraulic and physical properties such as transmissivity and porosity (which determine transport velocities) to be adequately represented by averages. Consequently, one should not expect the simulations to accurately predict detailed distributions of contaminants in the Maynardville, but only to give a general sense of the extent of plume migration.

The second potential limitation is that the model is two-dimensional. This implies that aquifer properties and concentrations are uniform with depth and, consequently, that groundwater flow and contaminant transport are entirely horizontal. Aquifer properties at the Oil Landfarm WMA are not uniform with depth, nor are contaminant concentrations, as indicated by monitoring well data. In addition, monitoring well data and preliminary modeling (Bailey 1988) indicate vertical components to groundwater flow, especially in the vicinity of Bear Creek. However, given other uncertainties in the simulations and the paucity of data on the vertical components of flow, a two-dimensional treatment, with aquifer properties averaged or integrated over the vertical dimension appears to be adequate for the purpose of risk assessment.

The third assumption in the model of potential significance to the simulations is that contaminants in the aquifer are transported in the aqueous phase. This assumption is related to the problem of vertical flow but is treated separately here because it has additional ramifications. As discussed in Sect. 7, the VOCs are only slightly water soluble and are much denser than water; large volumes of these compounds have, therefore, probably entered the aquifer as nonaqueous phase liquids (NAPLs). The MOC model clearly cannot describe contaminant transport in an NAPL column. However, the column itself is relatively immobile in the horizontal plane, and groundwater contamination away from the source area can be adequately modeled by MOC. The consequences of the probable long-term NAPL source are discussed in Sect. 7.

## A.2 MODEL CALIBRATION

### A.2.1 Introduction

Model calibration entails customizing the generic MOC model to treat the aquifer and the contaminants under study. The first step in calibration is to input estimates of several hydrogeological parameters, which are derived from data from field measurements. The measured hydraulic head distribution over the area to be modeled is then reproduced in the model simulations by systematically adjusting these parameters (within limits consistent with the data) and specifying appropriate boundary conditions. When this step is complete, initial estimates of porosity, dispersivity, contaminant properties, and a contaminant mass input rate (i.e., the source term) are made and input to the model. Another series of simulations is then run, and these parameters are adjusted (again within limits consistent with the data) until measured contaminant concentrations at some time are adequately reproduced. The solution produced this way is not unique, but it is consistent with all available data. The choices of aquifer parameters, contaminant properties and source terms are discussed in the next two sections; calibration results are shown in the final section of this appendix.

## A.2.2 Aquifer Parameters

To perform hydrological and solute transport simulations, the dimensions, boundary conditions, and physical properties of the aquifer and the properties of the contaminants must be input to the model. Table A.1 lists the values of the aquifer and contaminant parameters used in the Oil Landfarm simulations. The area of the model aquifer is chosen to be large enough that the boundaries in the transport direction do not significantly affect the flow regime or contaminant transport. The dimensions used in this study are $6000 \times 4000$ ft, giving a total area of $2.4 \times 10^7$ ft$^2$ (Fig. A.1). The model aquifer is divided into 384 square finite-difference cells, each 250 ft on a side. Because calculated heads and concentrations are averaged over the dimensions of a cell, the size of a cell corresponds to the spatial resolution of the simulation results.

### Table A.1. Model parameters

| | |
|---|---|
| Aquifer area | $6000 \times 4000$ ft |
| Cell area | $250 \times 250$ ft |
| Aquifer thickness | 600 ft |
| Recharge | $2.0 \times 10^{-8}$ ft/s |
| Leakance per Bear Creek Node | $4.0 \times 10^{-10}$/s |
| Transmissivity in all units except Maynardville Limestone | |
|   Perpendicular strike | $7.2 \times 10^{-4}$ ft$^2$/s |
|   Parallel strike | $3.6 \times 10^{-3}$ ft$^2$/s |
| Transmissivity in Maynardville Limestone | |
|   Perpendicular strike | $2.2 \times 10^{-3}$ ft$^2$/s |
|   Parallel strike | $1.1 \times 10^{-1}$ ft$^2$/s |
| Effective porosity | 0.02 |
| Longitudinal dispersivity | 30 ft |
| Transverse dispersivity | 10 ft |
| Aquifer bulk density | $2.5 \times 10^6$ g/m$^3$ |
| Organic carbon fraction | $1.0 \times 10^{-4}$ g carbon/g dry soil |
| | |
| Retardation factors | |
|   Tetrachloroethene | 4.1 |
|   Trichloroethene | 2.9 |
|   1,1-Dichloroethene | 2.1 |

The depth to which significant contamination occurs in the aquifer at the Oil Landfarm WMA is not well characterized. However, water quality data indicate that serious contamination extends to greater than 200 ft in both the Nolichucky and the Maynardville formations at the site (The deepest well at the site is screened at 245 ft.) In addition, data from the Bear Creek Burial Grounds site, located less than a mile down Bear Creek Valley, indicate high VOC concentrations in a well over 500 ft deep in the Nolichucky. The thickness of the aquifer is chosen to be a uniform 600 ft to cover the maximum vertical extent of contamination suggested by these data. Measured hydraulic conductivities in the Nolichucky at depths over 500 ft are very low, and 600 ft is probably near the upper limit on
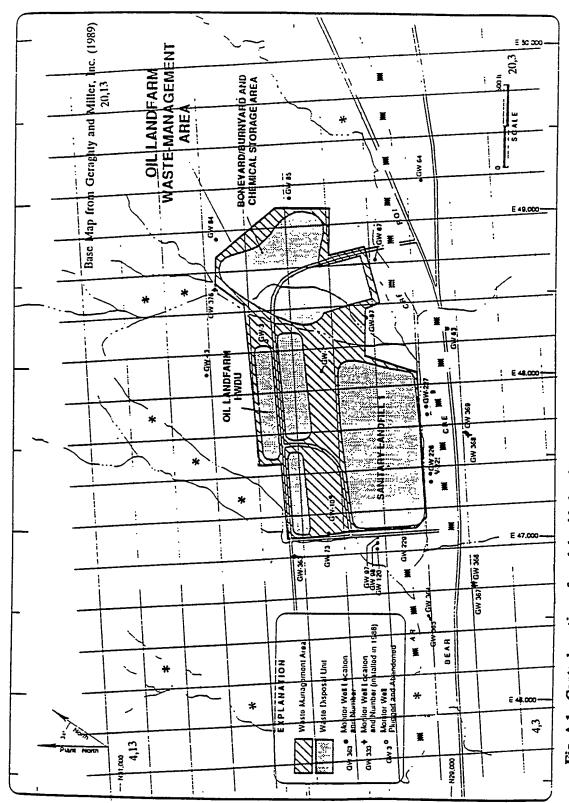
Fig. A.1. Central portion of model grid showing east-west cells 4 through 20, north-south cells 3 through 13, and boundary conditions. * = constant head node. ▧ = leakance node. Outside boundaries (not shown) are zero flux boundaries.

aquifer thickness. The unconsolidated and consolidated zones are treated together as a single aquifer. The vadose zone is quite thin compared to the aquifer and is considered negligible.

Two types of boundary conditions are specified for the simulations: prescribed flux and constant head (Fig. A.1). The outermost cells of the finite-difference grid are automatically defined in the model as boundaries across which the fluid and solute fluxes are zero. These no-flow boundaries are appropriate for the northern and southern edges of the modeled area, which correspond to Pine Ridge and Chestnut Ridge, respectively. They are not appropriate for the western and eastern boundaries, however, where there are no natural barriers to flow. This problem is handled by putting these boundaries at distances from the source area that are great enough that they do not influence fluid flow or solute transport.

Recharge to the aquifer is another prescribed flux boundary condition. The areal distribution of recharge in Bear Creek Valley is not known, although model studies by Bailey (1988) suggest recharge occurs mainly on Pine Ridge and Chestnut Ridge. Because of the two-dimensional nature of this study, we consider it adequate to use an average constant recharge over the area of the aquifer. Bear Creek Valley water budget simulations for the years 1983 and 1984 (using the Terrestrial Ecosystem Hydrology Model (TEHM)) suggest a value of recharge of about $5 \times 10^{-8}$ ft/s (19 in/year) (Geraghty and Miller, Inc. 1985). Later work referred to in Bailey (1988) indicates that recharge may be less than half that value. The value chosen for this study, $2 \times 10^{-8}$ ft/s (7.57 in./year), is based on Bailey (1988) and estimates of discharge to Bear Creek (see below). More recently, G. K. Moore (1988) suggests that recharge to the deep aquifer may be even less, on the order of 1 in./year.

Bear Creek and the shallow cavity system are modeled as a combined fluid sink using leakance terms in model cells corresponding to the location of the creek. The value of leakance necessary to reproduce measured hydraulic heads on the valley floor is very sensitive to the value chosen for recharge, and satisfactory fits to head data can be made for various combinations of these two parameters. Using the recharge value discussed above and a leakance value per node of $4 \times 10^{-10}$/s gives a total flow out all Bear Creek nodes of approximately 0.5 ft³/s or $9.1 \times 10^{-5}$ ft³/s/ft of stream. The simulated discharge is consistent with the average annual groundwater input to the reach of Bear Creek above Tributary 3 of $9.3 \times 10^{-5}$ ft³/s/ft of stream calculated from surface flow data (Martin Marietta Energy Systems, Inc. 1985a). (The reach of Bear Creek south of the Oil Landfarm WMA is losing during much of the year and could not be used to estimate discharge.) The flows in Tributaries 3 and 4 are generally about 10 times less than flows in Bear Creek, and the contribution of groundwater to these flows is uncertain. Because they are not expected to act as significant sinks, the tributaries are not explicitly modeled.

Constant head boundaries are used in model cell locations on Chestnut Ridge and Pine Ridge to drive the flow system and to shape the water table across the area. Because the flow simulations are steady-state, the hydraulic head contours do not change with time. The contours used in the simulations are calibrated to data for 1988 (Geraghty and Miller, Inc. 1989). These contours are similar to those measured over the past few years (Bechtel National, Inc. 1983; Bechtel National, Inc. 1984a; Geraghty and Miller, Inc. 1985; Geraghty and Miller, Inc. 1987a; Geraghty and Miller, Inc. 1989). Although heads fluctuate seasonally, the pattern of change is similar in all wells, and hydraulic gradients remain relatively constant.

The physical properties of the aquifer, which must be specified in the model, are transmissivity, effective porosity, and dispersivity. The values for these parameters are derived

from aquifer tests which have been performed over the past few years. Although other lithologies underlie parts of the Oil Landfarm WMA, most available test data are for the Nolichucky Shale.

Because the transmissivity in the Nolichucky varies with depth, an initial estimate for an average value for use in the two-dimensional simulations is obtained by determining the hydraulic conductivity, $K$ (ft/s), as a function of depth, $z$ (ft), from the results of recent tests in wells in the Nolichucky (Geraghty and Miller 1987e) and integrating the result over the thickness of the aquifer. Figure A.2 shows ln $K$ vs $z$ with a best-fit line. The correlation coefficient for this line is 0.96; however, this excellent correlation is somewhat forced because there are no data points in the middle of the depth range. The functional relationship between $K$ and $z$ implied by the best-fit line is:

$$K = e^{-(11.1 + 0.019z)} .$$

<div align="right">(A.1)</div>

Integrating this expression over an aquifer depth of 600 ft gives an average transmissivity of $7.9 \times 10^{-4}$ ft$^2$/s. This value is in the range of values implied by other pump tests (Law Engineering Testing Company 1983; Geraghty and Miller, Inc. 1987d; Geraghty and Miller 1985; Lozier, Spiers, and Pearson 1987). Test data also clearly indicate that the Nolichucky is strongly anisotropic with transmissivity along bedding plane strike ($T_{xx}$) a factor of 5 to 20 larger than transmissivity perpendicular to strike ($T_{yy}$). (For the simulations, the axes of the finite-difference grid are oriented parallel to the principal directions of transmissivity (i.e., parallel and perpendicular to strike). Initially, the calculated value of transmissivity was assigned to $T_{xx}$, consistent with the idea that pump tests measure chiefly transmissivity in this direction. During calibration, however, we found that transmissivities about a factor of 5 above initial estimates were necessary to avoid mounding of the water table, and best fits to hydraulic head data were obtained with $T_{xx}$ equal to $3.6 \times 10^{-3}$ and $T_{yy}$ equal to $7.2 \times 10^{-4}$, values still within the range of measurements. Some transmissivity data are available for the Maryville Limestone and the Knox Dolomite from areas outside Bear Creek Valley; however, because significant transport does not occur in these formations at this site, we lump them with the Nolichucky for convenience and assign the same transmissivities to them.

A significant portion of the solute transport in the simulations of this site takes place through the region representing the Maynardville Limestone, and test data suggest that the transmissivity of the Maynardville may be an order of magnitude or more greater than that of the Nolichucky (Bechtel National, Inc. 1984b; Lozier, Spiers, and Pearson 1987). The model allows one to specify different transmissivities at different locations in the aquifer (heterogeneous aquifer). For the present simulations, the transmissivity used for the Maynardville is three times that of the Nolichucky. Use of a factor of 3 rather than 10 or more is based on the expectation that the cavity system does not extend to the full 600 ft of the model aquifer. An attempt is made to account for bedding plane dip by varying the actual values assigned to specific model cells: nodes (2,5) to (23,5) are assigned a full $T_{yy}$ of $2.2 \times 10^{-3}$, and nodes (2,6) to (23,6) are assigned a smaller $T_{yy}$ of $1.4 \times 10^{-3}$ because the Nolichucky dips below this area. Transmissivities parallel strike are five times those perpendicular to strike.

Estimates of effective porosity for formations of the Conasauga and the unconsolidated zone range from 0.002 to 0.1 (Law Engineering Testing Company 1983; Geraghty and Miller 1985; Bechtel National, Inc. 1984b; Lozier, Spiers, and Pearson 1987), with the upper limits
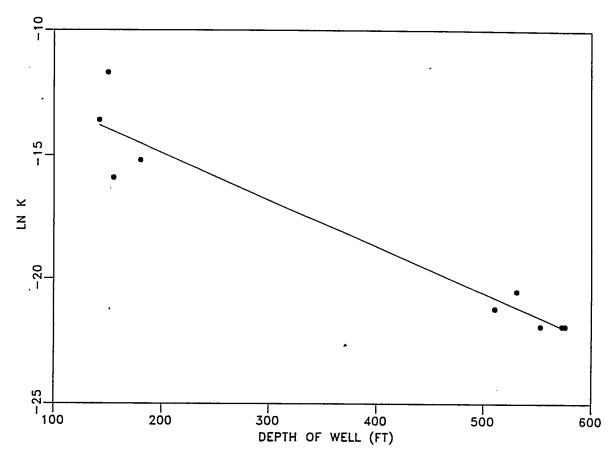
**Fig. A.2.** Hydraulic conductivity as a function of depth. Data from Geraghty and Miller, Inc. 1987e.

associated with the unconsolidated material. The model allows for only one value of effective porosity. The value used in the simulations (0.02) is a roughly weighted average over the depth of the aquifer, assuming that effective porosity decreases with depth.

Results of tracer tests in the Nolichucky (Law Engineering Testing Company 1983; Lozier, Spiers, and Pearson 1987) give a longitudinal dispersivity of 10 ft. However, in the simulations, a longitudinal dispersivity of 30 ft better reproduces existing contaminant distributions. The necessity of using larger than measured or predicted dispersivities to model large contaminant plumes is a common occurrence in modeling studies and may indicate that dispersion is a scale dependent phenomenon (Pickens and Grisak 1981). A longitudinal dispersivity of 30 ft is, therefore, used in the simulations. The only data on the ratio of the transverse to the longitudinal dispersivity for the Bear Creek Valley aquifer are from Lozier, Spiers, and Pearson (1987) who report a value of 1. Values reported in the literature range from 0.1 to 0.3. Simulation results are not sensitive to variations over this range, so a value of 0.3 is, somewhat arbitrarily, used.

## A.2.3 Retardation Factors

Contaminants adsorb to some extent onto aquifer matrix materials, and their movement through the aquifer is inhibited. The model represents this process by applying a retardation factor to the transport calculations. The transport velocity of a sorbing solute is equal to the seepage velocity of the groundwater divided by the retardation factor. The retardation factor is calculated from the following expression (Roberts et al. 1985):

$$R = 1 + K_d \, r/n \qquad (A.2)$$

where $R$ is the retardation factor, $r$ is the bulk density of the aquifer matrix (g/m$^3$), $n$ is the porosity, and $K_d$ is the soil/water distribution coefficient (m$^3$/g). The bulk density used in the simulations (2.5 $\times$ 10$^6$ g/m$^3$) is an average value for Paleozoic shales at depths from 1000 to 5000 ft (Dobrin 1976). Porosity is assumed equal to effective porosity, the choice of which was discussed earlier. The distribution coefficient is calculated from the following expression (Karickhoff et al. 1979):

$$K_d = 6.3 \times 10^{-7} f_{oc} \, (K_{ow}) \, , \qquad (A.3)$$

where $K_d$ is the soil/water distribution coefficient (m$^3$/g), $f_{oc}$ is the fraction of organic carbon in the matrix (g carbon/g dry soil), and $K_{ow}$ is the octanol-water partition coefficient (molar concentration basis). Values of $K_{ow}$ for PCE and TCE are obtained from the *Superfund Public Health Evaluation Manual* (EPA 1986). The $K_{ow}$ used for 1,1-DCE is from Arthur D. Little, Inc. (1985) (Table 7.1). The biggest uncertainty involved in the calculation of retardation factors is the estimation of the organic carbon fraction. Measured values in soils from Bear Creek Valley range from 0.0 to 0.011, with 0.001 the most often reported value (Martin Marietta Energy Systems, Inc. 1985b). The average value for the aquifer as a whole is undoubtedly much smaller than the reported soil values. Schwarzenbach and Giger (1985) report $f_{oc}$ values from <0.0001 to 0.0008 for mineral surfaces and organic-poor aquifers. Based on these data and the expectation of very low organic carbon content for the aquifer, a value for $f_{oc}$ of 0.0001 is chosen for the simulations. This value gives retardation factors in a reasonable range (Table A.1) and produces adequate calibration results.

## A.2.4 Source Terms

### A.2.4.1 Introduction

In the Oil Landfarm WMA simulations, contaminant input to the aquifer is represented by assigning concentrations to annual recharge in the model cells corresponding to source locations. A source term in the model is then the product of the recharge (ft/s), the cell area ($ft^2$), and the concentration of the contaminant $\mu g/ft^3$ (i.e., the mass of contaminant per unit time input to a cell). For the no action, capping-with-sources and recovery-with-sources simulations, the source terms are one of the most important model parameters to be specified. They determine both the concentrations in the cores of the plumes and the persistence of the plumes with time. For the Oil Landfarm WMA simulations, historical and other data allow reasonable inferences of the locations of contaminant sources. As we will discuss below, however, both the magnitudes and durations of the source terms are poorly constrained, leading to large uncertainties in concentrations in simulations involving source terms and an inability to predict persistence.

### A.2.4.2 Locations of contaminant sources

Current groundwater contamination patterns suggest at least three separate sources of contaminants at the Oil Landfarm WMA (Sect. 4). The locations of model contaminant sources are based on historical data about disposal practices at the site, current groundwater and soil contamination patterns, and geophysical data. VOC-containing solvents are known to have been poured into unlined trenches in the southwest corner of the Boneyard and in the Burnyard, and they may also have been disposed of in the Chemical Storage Area. The solvents deposited in these areas are almost certainly the source of contamination to GW-87, which is located immediately downgradient and contains high levels of PCE and TCE. Electromagnetic terrain conductivity measurements (Weston Geophysical Corporation 1984) show anomalously high conductivity in this vicinity and also strongly suggest a contaminant source in the area. In the simulations a source term for PCE and TCE, labeled I in Fig. A.3, represents this source.

VOC-containing oils were cultivated into the soil in the Oil Landfarm plots and are without doubt the source of contamination to wells located south of the plots (e.g., GW-7). VOC levels in soils are almost two orders of magnitude higher in the southeast group of plots than in the southwest group (Bechtel National, Inc. 1983) and are below detection limits in the northern plots, suggesting that a major source of groundwater contamination associated with the plots is in or below the southeast group. Current VOC concentrations in groundwater support this view: they are greatest in wells south of the southeast plots (e.g., GW-7), which contains relatively high levels of PCE and TCE, and very low in wells south of the southwest plots. Based on these data, the simulations use source terms in two cells corresponding to the southeast plots to represent the Oil Landfarm-derived PCE and TCE sources; these are labeled II and III in Fig. A.3.

GW-3, located just northeast of the northern Oil Landfarm plots, is the only well that contains high concentrations of 1,1,1-trichloroethane, 1,1-dichloroethane, and 1,1-DCE. 1,1-Dichloroethane and 1,1-DCE are probably transformation products of 1,1,1-trichloroethane (see Sect. 7), and the observed assemblage probably results from an isolated disposal of 1,1,1-trichloroethane-contaminated material probably in the northern Oil Landfarm disposal plots.
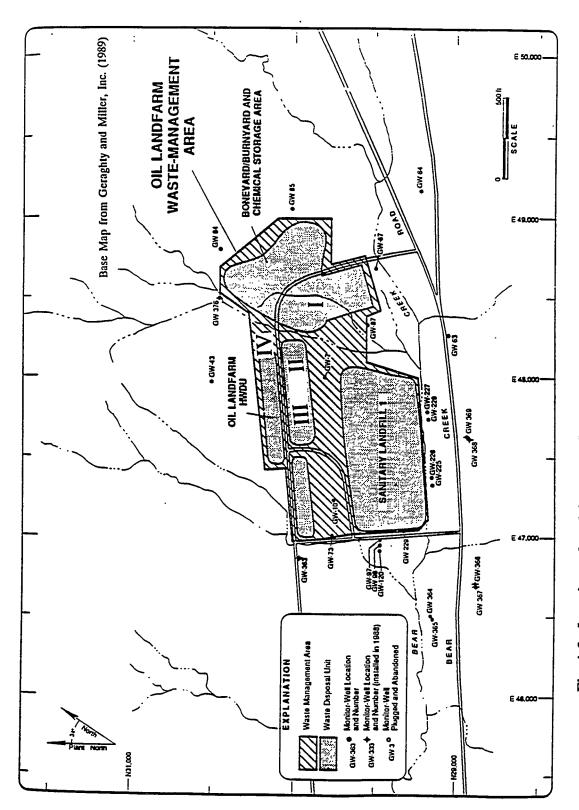
Fig. A.3. Locations of model sources. I and II–PCE and TCE. III–PCE. IV–1,1-DCE.

Because of the distribution and size of model cells, the source term is assigned directly to the model cell containing GW-3 (labeled IV in Fig. A.3).

### A.2.4.3 Source-term magnitudes and duration

There are no data on the amounts of VOCs disposed of at the Oil Landfarm WMA from which to constrain source-term magnitudes. Consequently, pre-1988 source-term magnitudes are based almost entirely on best fits to measured concentration data for GW-87, GW-7, and GW-3 (see also Sect. A.2.5.2). The appropriate duration of the pre-1988 Oil Landfarm plot sources (II and III) is about 15 years, because disposal in these plots began in 1973. The appropriate duration for source I is more difficult to estimate. Disposal in the Boneyard and the Burnyard began in 1943 for both areas and ended in 1970 and 1968, respectively. Consequently, these sites could have been inputing contaminants to the aquifer for about 18 to 45 years. If source I is associated with these areas, the appropriate duration of the pre-1988 source term is in this range. If source I is associated with the Chemical Storage Area, its durations should be about 12 years; this area operated from 1975 to 1981. Because the northern Oil Landfarm disposal plots are probably the source for the contamination in GW-3, the appropriate duration for this source is 15 years or less.

In practice the magnitudes of the source terms necessary to reproduce measured 1988 concentrations in GW-87, GW-7, and GW-3 are essentially the same for any simulation time. However, longer simulation times require higher retardation factors to avoid seriously overestimating the extent of 1988 plumes (as judged by low concentrations in wells on the periphery of the WMA). For example, calibrating to 1988 GW-87 concentrations with source term I over a 45-year period requires a retardation factor 3 times that used to calibrate over a 15-year period. Based on the properties of the site and experience with calibration in the Nolichucky at the Bear Creek Burial Grounds, where source-term durations were fairly well constrained, the retardation factors corresponding to about 15-year calibration times appear to be more realistic. Consequently, 15 years is used as the pre-1988 PCE and TCE source-term duration for source I as well as source II and source III. Using a retardation factor for 1,1-DCE consistent with those used for TCE and PCE requires a pre-1988 source-term duration of 10 years, which is plausible since 1,1-DCE is probably a transformation product.

Ideally, post-1988 source terms should decrease with time, but, there are no data from which to infer the form or rate of decrease. As a worst-case scenario, therefore, this study uses post-1988 source terms which are constant and the same magnitude as pre-1988 source terms. These source terms input contaminant mass to the aquifer for the full 115 (for PCE and TCE) or 110 (for 1,1-DCE) (i.e., calibration to 1988 plus 100 years) years of simulation. This approach almost certainly overestimates post-1988 simulated no-action concentrations, but does not imply altogether unreasonable amounts of contaminants (e.g., PCE source terms running for 115 years input less than five 55-gal drums of the compound to the aquifer). Because of the large uncertainty in source terms, predicted concentrations should be considered within an order of magnitude. Source-term parameters and total mass and volume of free product input to the aquifer for PCE, TCE, and 1,1-DCE are shown in Table A.2.

Table A.2. VOC source terms and mass input

| Contaminant | Source[a] | | | |
| | I | II | III | IV |
|---|---|---|---|---|
| **Tetrachloroethene** | | | | |
| Source term ($\mu g/s$) | $4.3 \times 10^2$ | $1.4 \times 10^2$ | $2.1$ | |
| Flow rate (L/s) | $3.5 \times 10^{-2}$ | $3.5 \times 10^{-2}$ | $3.5 \times 10^{-2}$ | |
| Concentration ($\mu/L$) | $1.2 \times 10^4$ | $4.0 \times 10^3$ | $6.0 \times 10^1$ | |
| Mass input in 115 years ($\mu g$) | $1.5 \times 10^{12}$ | $5.1 \times 10^{11}$ | $7.6 \times 10^9$ | |
| Volume input in 115 years (gal) | $2.4 \times 10^2$ | $8.3 \times 10^1$ | $1.2$ | |
| | | | | |
| **Trichloroethene** | | | | |
| Source term ($\mu/s$) | $4.6 \times 10^2$ | $1.8 \times 10^1$ | | |
| Flow rate (L/s) | $3.5 \times 10^{-2}$ | $3.5 \times 10^{-2}$ | | |
| Concentration ($\mu/L$) | $1.3 \times 10^4$ | $5.0 \times 10^2$ | | |
| Mass input in 115 years ($\mu g$) | $1.7 \times 10^{12}$ | $6.4 \times 10^{10}$ | | |
| Volume input in 115 years (gal) | $3.0 \times 10^2$ | $1.2 \times 10^1$ | | |
| | | | | |
| **1,1-Dichloroethene** | | | | |
| Source term ($\mu/s$) | | | | $4.6 \times 10^2$ |
| Flow rate (L/s) | | | | $3.5 \times 10^{-2}$ |
| Concentration ($\mu/L$) | | | | $1.3 \times 10^4$ |
| Mass input in 110 years ($\mu g$) | | | | $1.6 \times 10^{12}$ |
| Volume input in 110 years (gal) | | | | $3.5 \times 10^2$ |

[a]See Fig. A.3 for source locations.

## A.2.5 Calibration Results

### A.2.5.1 Water table

The model water table for the Oil Landfarm WMA is calibrated to measured water level elevations for 1988, which are reported in Geraghty and Miller, Inc. (1989). Water level elevations were not measured for all wells in any single quarter, but more wells were measured in the first quarter (April 22 to April 27) than any other quarter. Consequently, the model water table is calibrated primarily to measurements for the first quarter. Water levels were measured in several new wells in the Maynardville along the floor of Bear Creek Valley in the third and fourth quarters, and the model water table is calibrated to fourth-quarter measurements for these wells. No trend is discernible between first- and fourth-quarter water levels in other wells in the Maynardville, which were measured both quarters: fourth-quarter water levels are higher than first-quarter levels in some wells and lower than first-quarter levels in others. Table A.3 shows a comparison between measured and modeled water levels at the Oil Landfarm WMA. For well clusters in a single-model cell, modeled levels are compared to the average value for wells in the cluster. Differences between measured and modeled elevations are generally about 1 to 3 ft, with over 60% of the wells showing differences of less than 2 ft. A major discrepancy seems to exist between measured and modeled levels for GW-43; however, the reported measured value appears questionable, because values reported in other years are consistently 10 ft or more lower than the 1988 value. Figure A.4 illustrates the simulated water table contours.

Table A.3. Water table calibration results

| Monitoring well | Water level elevation (ft, MSL) | | |
|---|---|---|---|
| | 1988 Measured[a] | Simulated | Simulated minus measured |
| GW-3 | 966.3 | 969 | 2.7 |
| GW-7 | 956.5 | 958 | 1.5 |
| GW-10 | 949.0 | 948 | -1.0 |
| GW-43 | 999.6 | 985 | -14.6 |
| GW-63 | 933.2 | 939 | 5.8 |
| GW-64 | 955.8 | 954 | -1.8 |
| GW-67 | 952.2 | 947 | -5.2 |
| GW-73 | 946.4 | 945 | -1.4 |
| GW-84 | 982.6 | 982 | -0.6 |
| GW-85 | 967.3 | 969 | 1.7 |
| GW-87 | 953.0 | 951 | -2.0 |
| | | | |
| GW-97 | 935.0 | | |
| GW-98 | 934.5 | | |
| GW-120 | 942.2 | | |
| Average | 937.2 | 936 | -1.2 |
| | | | |
| GW-225 | 925.1 | | |
| GW-226 | 930.4 | | |
| Average | 927.8 | 928 | 0.3 |
| | | | |
| GW-227 | 933.9 | | |
| GW-228 | 932.9 | | |
| Average | 933.4 | 933 | -0.4 |
| | | | |
| GW-229 | 928.3 | 928 | -0.3 |
| GW-363 | 951.2 | 950 | -1.2 |
| | | | |
| GW-364 | 916.4 | | |
| GW-365 | 926.0 | | |
| Average | 921.2 | 919 | -2.2 |
| | | | |
| GW-366 | 917.7 | | |
| GW-367 | 919.2 | | |
| Average | 918.5 | 924 | 5.5 |
| | | | |
| GW-368 | 940.6 | | |
| GW-369 | 921.9 | | |
| Average | 931.3 | 930 | -1.3 |

[a]GW-3 to GW-229 1st quarter data. GW-363 to GW-369 4th quarter data. *Source:* Geraghty and Miller, Inc. 1989.
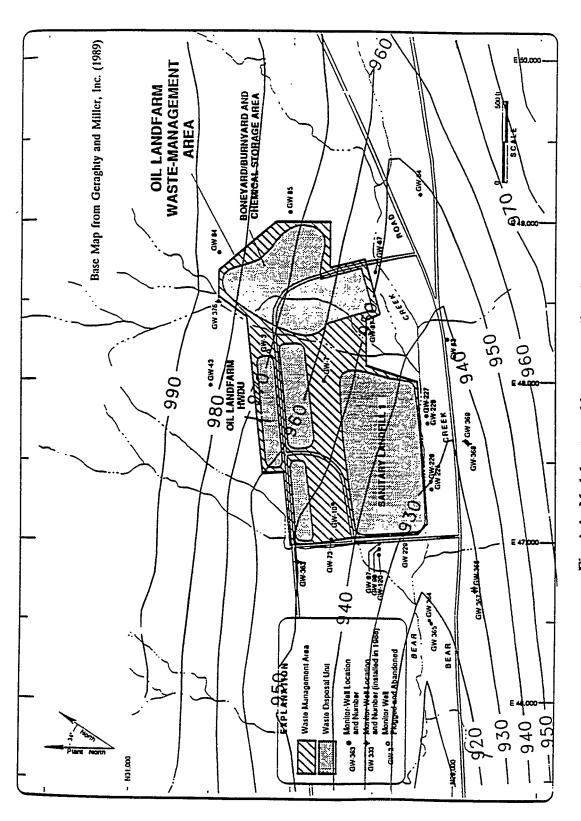
140



Fig. A.4. Model water-table contours (ft, msl).

## A.2.5.2 Contaminant transport

High concentrations of contaminants occur only in a few wells at the site (Figs. 4.2–4.4). This spatial distribution of contaminants does not allow a really satisfactory transport calibration. The quality of the calibration must be judged primarily by best fits to GW-87 and GW-7 for PCE and TCE, and GW-3 for 1,1-DCE. The only other calibration criterion is that modeled concentrations at the locations of wells around the periphery of the WMA (e.g., GW-10, GW-225, GW-120) be small, as observed in the well data.

Model concentrations for PCE and TCE in GW-87 are calibrated to 4-year averages of measured concentrations for 1984, 1986, 1987, and 1988 (VOCs were not sampled in 1985). We chose to calibrate to 4-year average concentrations for this well rather than to 1988 concentrations because, though there is no trend of decreasing concentrations with time over the 5-year period, concentrations measured for 1988 are significantly lower than in other years. Four-year averages thus appear to better represent concentrations in this well than do 1988 concentrations. Concentrations of 1,1-DCE in GW-3 were measured only in 1984 and 1988, so it is not possible to evaluate trends in the data. Model concentrations in GW-3 are, however, somewhat arbitrarily calibrated to 2-year averages for the years 1984 and 1988. Model concentrations for all other wells are calibrated to average measured concentrations for 1988. If two or more wells fall in the same model cell, model concentrations are calibrated to the average of concentrations measured in the wells.

Tables A.4–A.6 show comparisons of measured and modeled concentrations for PCE, TCE, and 1,1-DCE, respectively. For PCE the model reproduces the measured data in GW-87 and GW-7 quite well and produces only very small concentrations around the periphery of the site, as desired. For TCE the model reproduces GW-87 concentrations but seriously overestimates GW-7 concentrations. However, concentration gradients are very steep across the model cell containing GW-7, and the concentration in the adjacent cell to the southwest is only 39 $\mu$g/L, which is reasonably consistent with the measured GW-7 concentration of 16 $\mu$g/L. Model concentrations around the periphery of the WMA are small, consistent with the available data. The data for wells in the Maynardville, which appear to be contaminated from another source upstream from the Oil Landfarm WMA (see Sect. 4), were not considered in the calibration.

For 1,1-DCE the model reproduces the measured concentration in GW-3 but seriously overestimates the concentration in GW-7. In this case the discrepancy cannot be attributed to steep concentration gradients across model cells. The model may in fact be overestimating 1,1-DCE concentrations across the entire area between GW-3 and the western and southern boundaries of the WMA. However, there are no wells other than GW-7 with which to compare model predictions. Predicted concentrations around the periphery of the site, however, approximate the measured data.

Table A.4. Tetrachloroethene calibration results

| Monitoring well | Concentration ($\mu g/L$) | |
|---|---|---|
| | 1988 Average measured[a] | Simulated |
| GW-3 | 0 | 22 |
| GW-7 | 71 | 82 |
| GW-10 | 5 | 9 |
| GW-43 | 1 | |
| GW-63 | 1 | |
| GW-64 | 1 | |
| GW-67 | 0 | |
| GW-73 | 1 | |
| GW-84 | 1 | 1 |
| GW-85 | 0 | 1 |
| GW-87 | 395 (696[b]) | 695 |
| GW-97 | 1 | |
| GW-98 | 0 | |
| GW-120 | 1 | |
| Average | 1 | 0 |
| GW-225 | 5 | |
| GW-226 | 2 | |
| Average | 4 | 0 |
| GW-227 | 2 | |
| GW-228 | 4 | |
| Average | 3 | 1 |
| GW-229 | 0 | 0 |
| GW-363 | 0 | 1 |
| GW-364 | 1 | |
| GW-365 | 19 | |
| Average | 10 | 0 |
| GW-366 | 3 | |
| GW-367 | 3 | |
| Average | 3 | 0 |
| GW-368 | 1 | |
| GW-369 | 1 | |
| Average | 1 | 1 |

[a]Source: Geraghty and Miller, Inc. 1989.

[b]Four-year average. See text for explanation. Sources: Geraghty and Miller, Inc. (1989), (1988), (1987a); Martin Marietta Energy Systems, Inc. (1985b).

Table A.5. Trichloroethene calibration results

| Monitoring well | Concentration (μg/L) | |
| --- | --- | --- |
| | 1988 Average measured[a] | Simulated |
| GW-3 | 0 | 25 |
| GW-7 | 16 | 200 |
| GW-10 | 5 | 8 |
| GW-43 | 0 | 0 |
| GW-63 | 44 | b |
| GW-64 | 165 | b |
| GW-67 | 3 | b |
| GW-73 | 1 | 1 |
| GW-84 | 1 | 0 |
| GW-85 | 0 | 1 |
| GW-87 | 315 (500[c]) | 512 |
| GW-97 | 0 | |
| GW-98 | 9 | |
| GW-120 | 1 | |
| Average | 3 | 1 |
| GW-225 | 418 | |
| GW-226 | 15 | |
| Average | b | b |
| GW-227 | 37 | |
| GW-228 | 77 | |
| Average | b | b |
| GW-229 | 1 | b |
| GW-363 | 3 | b |
| GW-364 | 26 | |
| GW-365 | 84 | |
| Average | b | b |
| GW-366 | 12 | |
| GW-367 | 26 | |
| Average | b | b |
| GW-368 | 125 | |
| GW-369 | 110 | |
| Average | b | b |

[a]Source: Geraghty and Miller, Inc. (1989).
[b]Wells in the Maynardville not used for calibration. See text for explanation.
[c]Four-year average. See text for explanation. Sources: Geraghty and Miller, Inc. (1989), (1988), 1987a; Martin Marietta Energy Systems, Inc. 1985b.

Table A.6. 1,1-Dichloroethene calibration results

| Monitoring well | Concentration ($\mu g/L$) | |
| --- | --- | --- |
| | 1988 Average measured[a] | Simulated |
| GW-3 | 620 (735)[b] | 727 |
| GW-7 | 7 | 573 |
| GW-10 | 5 | 23 |
| GW-43 | 0 | 1 |
| GW-63 | 0 | 0 |
| GW-64 | 1 | 0 |
| GW-67 | 1 | 0 |
| GW-73 | 1 | 3 |
| GW-84 | 0 | 1 |
| GW-85 | 0 | 0 |
| GW-87 | 4 | 9 |
| | | |
| GW-97 | 0 | |
| GW-98 | 0 | |
| GW-120 | 0 | |
| Average | 0 | 0 |
| | | |
| GW-225 | 5 | |
| GW-226 | 0 | |
| Average | 3 | 1 |
| | | |
| GW-227 | 0 | |
| GW-228 | 0 | |
| Average | 0 | 0 |
| | | |
| GW-229 | 0 | 1 |
| GW-363 | 0 | 1 |
| | | |
| GW-364 | 9 | |
| GW-365 | 27 | |
| Average | 18 | 0 |
| | | |
| GW-366 | 0 | |
| GW-367 | 1 | |
| Average | 1 | 0 |
| | | |
| GW-368 | 1 | |
| GW-369 | 1 | |
| Average | 1 | 0 |

[a]Source: Geraghty and Miller, Inc. 1989.

[b]Four-year average. See text for explanation. Sources: Geraghty and Miller, Inc. (1989); Martin Marietta Energy Systems, Inc. (1985b).

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